

# Ancillary Benefits and Costs of Greenhouse Gas Mitigation

ENVIRONMENT



WRI



INTERGOVERNMENTAL PANEL ON CLIMATE CHANGE (IPCC)



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# **Ancillary Benefits and Costs of Greenhouse Gas Mitigation**

Proceedings of an  
IPCC Co-Sponsored Workshop,  
held on 27-29 March 2000,  
in Washington D.C.

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## FOREWORD

On 27 March 2000, I had the distinct pleasure to welcome to Resources for the Future (RFF) researchers from around the world convened to discuss the ancillary benefits and costs associated with measures to mitigate the growth of carbon dioxide and other greenhouse gases in the atmosphere. These ancillary effects (think of them as inadvertent consequences) are important to study because preliminary research shows that they might be of considerable quantitative and qualitative significance. For instance, controlling carbon dioxide emissions to reduce the likelihood or degree of global warming might, at the same time, reduce emissions of other pollutants that adversely affect human health and the environment. These “bonus” benefits ought rightly to be included in any accounting of the good that will be done by greenhouse gas mitigation. Similarly, greenhouse gas control policies can also have unexpected adverse consequences--and these, too, should be counted.

It gives me equal pleasure to welcome readers to a collection of interesting and important papers. Contributions are found from top-flight researchers from the United States and Europe, as well as from several of their counterparts from the developing world. Inclusion of developing country perspectives is significant for several reasons, not the least of which are that:

- i. many of the most significant adverse effects of global climate change are expected to occur in the developing world, principally because their relatively lower incomes will make adaptation more difficult; and
- ii. over time, emissions of carbon dioxide and other greenhouse gases from the developing world will gradually overtake those from the developed world.

Thus, on both the “cause” and “effect” dimensions, the developing world is increasingly important.

RFF was honoured to host this event and act as one of its co-sponsors. Other organisations playing a major role include the Intergovernmental Panel on Climate Change, the Organisation for Economic Co-operation and Development and the World Resources Institute. While I am recognising all those whose work was essential to the success of this workshop, I would like to single out Devra Lee Davis, Alan Krupnick, Gene McGlynn, who initiated, organised and conducted much of the meeting and also edited these proceedings.

**Paul R. Portney, Ph.D.**  
**Senior Fellow and President, Resources for the Future**

## ACKNOWLEDGEMENTS

The Proceedings and the Expert Workshop on Assessing the Ancillary Benefits and Costs of Greenhouse Gas Mitigation Strategies, 27-29 March 2000, was possible due to the generous support of the following organisations: Intergovernmental Panel on Climate Change (IPCC), Organisation for Economic Co-operation and Development (OECD), Resources for the Future (RFF), World Resources Institute (WRI), The Climate Institute, US Department of Energy, World Bank, W. Alton Jones, Statistics Norway, US Environmental Protection Agency, National Renewable Energy Laboratory (US) and the Rockefeller Family. Devra Davis (formerly with WRI and now with Carnegie Mellon University), Alan Krupnick (RFF), and Gene McGlynn (formerly with OECD), led preparations for the workshop and provided the leadership for the project resulting in completion of the proceedings. Jiahua Pan (IPCC), Jan Corfee-Morlot (OECD) and Bill Rhodes (USEPA) also provided useful guidance in the organisation of the meeting and completion of the proceedings. Paul Schwengels, Joel Scheraga and Myra Frazier (USEPA) also advised on the meeting as did lead authors from the IPCC Working Group II and III. Miranda Loh (WRI) assisted with organisation of the workshop and Andrew Gregory, Jane Kynaston, Kathleen Mechali, and Carolyn Sturgeon (OECD) provided technical assistance for production of the proceedings.

Internet Site: Further information on the Workshop and all of the papers in this volume may be found on the OECD's web site:

<http://www.oecd.org/env/cc>

Additional information on ancillary benefits may be found on the web sites for RFF, WRI and the IPCC (Working Group III):

<http://www.rff.org>

<http://www.wri.org>

<http://www.rivm.nl/env/int/ipcc>

<p>IPCC co-sponsorship does not imply IPCC endorsement or approval of these proceedings or any recommendations or conclusions contained herein. Neither the papers presented at the workshop/expert meeting nor the report of its proceedings have been subjected to IPCC review.</p>
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## PREFACE

The science of climate changes remains a matter of intense scientific debate and much public speculation. While the underlying science continues to evolve, efforts to figure out what are responsible policies to mitigate these potential impacts are also developing. In order to provide a rational means for choosing among policy alternatives to address this global problem, a number of national and international research and policy organisations are mounting major research efforts. The development of systematic methods for creating and assessing policies and programs to mitigate the direct and indirect impacts of climate change, and for estimating the costs and benefits of these policies, is the subject of an important and lively discussion in the technical and policy literature.

It is widely understood that policies devised to reduce or mitigate greenhouse gases (GHGs) can have positive and negative “ancillary effects” (one of many terms used to describe this phenomenon). Positive ancillary effects could include, for instance, reducing health-damaging emissions of conventional co-pollutants such as those tied with fossil fuels. Negative ancillary effects might include the morbidity and mortality from increased use of diesel fuels, which may, nevertheless, lower GHG emissions. The full array of these side effects of potential GHG mitigation policies is not always well understood, and consequently not integrated into policy-making.

On 27-29 March 2000, an international workshop to consider these issues in more detail was held, Intergovernmental Panel on Climate Change (IPCC), Organisation for Economic Co-operation and Development (OECD), Resources for the Future (RFF), World Resources Institute (WRI), The Climate Institute, US Department of Energy, World Bank, W. Alton Jones, Statistics Norway, US Environmental Protection Agency, National Renewable Energy Laboratory (US), the Rockefeller Family and the Economy and Environment Programme for South East Asia.

The workshop was designed to:

- provide information for the technical assessment efforts of the UN’s Intergovernmental Panel on Climate Change (IPCC), Third Assessment Report of Working Group III on Mitigation of Climate Change;
- integrate the quantification and consideration of ancillary effects of climate policies more clearly into the national and international policy process; and
- establish data gaps and research priorities.

This publication includes many of the papers presented at the workshop. Papers and presentations are also available at <http://www.oecd.org/env/cc>

This event brought together many of the leading experts on this topic to discuss their work and identify key issues for further analysis. The three days of the workshop covered methodologies and

frameworks, case studies, and links to policy-making. The papers in this volume of proceedings may have incorporated comments made at the Expert Workshop, but no formal review has been organised. The views expressed in this volume are those of the authors and not those of the co-sponsors. Discussants and participants contributed significantly to the Workshop, although their remarks are not included in this volume.

The workshop advanced understanding on common elements of an analytic framework for addressing this issue among the participants from more than 40 countries. The workshop also facilitated a dialogue between analysts in this field, as well as highlighting recent case studies from developed and developing countries. In particular, discussions emphasised the need to consider the complex role of national and multi-national institutions in affecting the level of ancillary effects. It also highlighted some continuing areas of debate, including valuation of health impacts and differences in approach between industrialised and developing countries. Much work remains to be done.

The workshop confirmed that positive and negative ancillary effects can be critical to the development of effective and efficient policy-making on GHG mitigation. Participants in the workshop considered data gaps and methodological issues relevant to improving the assessment of ancillary benefits and also laid out a research agenda that can be found at <http://www.wri.org>. The challenge remains how to incorporate current understanding into the evolving policy discussions and to better incorporate this complex issue into an already complex debate.

# ANCILLARY BENEFITS AND COSTS OF GREENHOUSE GAS MITIGATION

## AN OVERVIEW<sup>1</sup>

by Devra Lee DAVIS, Alan KRUPNICK and Gene McGLYNN

### 1. Introduction

Much of the debate over global climate change involves estimates of the direct costs of global climate change mitigation and the merits of various policies proposed to mitigate greenhouse gas emissions (GHG). Recently, this debate has broadened to include the issue of ancillary benefits and costs. It is generally understood that policies to reduce GHGs can have positive and negative “ancillary effects” on public health, ecosystems, land use, and materials and that such effects, if they can be monetized, can appropriately be subtracted from (or added to) mitigation cost to assess the social cost of such policies. Despite agreement that ancillary effects can be important, terminology to describe the effects and methods for estimation and valuation are in need of development and standardisation.

Whatever terminology is used to depict the indirect consequences of GHG mitigation policies, it is recognised that these effects can be constructive *or* harmful. Positive ancillary effects could result from, for instance, mitigation policies that reduce health- or environment-damaging emissions of conventional pollutants. Negative ancillary effects might result from those policies that increase health- or environmental damages, such as increased reliance on diesel fuels, which have lower greenhouse emissions than petrol but can increase health and environmental risks. These ancillary effects are not always well understood, and, until recently, have rarely been systematically quantified and valued. They are therefore seldom integrated into the development of GHG policies. Recent studies suggest that under some scenarios where baseline conditions include relatively high levels of pollution and inefficient abatement technologies, ancillary benefits of GHG mitigation policies can be of the same magnitude as the costs of proposed mitigation policies. Thus, the failure to consider ancillary effects may hamper the development of sound policy making.

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<sup>1</sup> The authors of this paper acknowledge the contributions of the authors who presented their papers at the workshop entitled “Ancillary Benefits and Costs of Greenhouse Gas Mitigation”. This overview paper draws extensively on the presentations of Luis Cifuentes, Richard Morgenstern, and David Pearce. It also draws on other papers presented in the workshop and where necessary to fill in the analytic gaps, on the wider literature on this topic. The paper is intended to be a broad overview of relevant issues rather than simply an overview of the workshop papers. The overview paper was developed after the meeting; thus it was not presented nor was it discussed at the meeting.

On 27-29 March 2000, an international workshop to consider these issues was held, in Washington D.C. The workshop was designed to provide information for the ongoing assessment efforts of the IPCC and other national and international agencies, to bring the ancillary benefits and costs of policies more clearly into the climate change debate, and to establish research priorities. This event brought together many of the leading experts on this topic to discuss their work and identify key issues for further analysis. The three days of the workshop covered methodologies and frameworks, case studies, and links to policy-making. While the workshop left many issues for further work, it advanced understanding on common elements of an analytic framework for addressing ancillary benefits and costs and facilitated a dialogue between analysts in this field. This summary report sets out some of the major issues addressed, areas of wide agreement and continuing controversies arising from the workshop and from the wider literature.

Section 2 discusses the basis for a common terminology and framework for analysis of ancillary effects, and sets out the key methodological issues involved. Section 3 provides a classification of potential ancillary effects. Section 4 then draws on these frameworks to examine existing empirical studies of ancillary effects. Section 5 discusses how ancillary effects analysis can impact on policy design and choice, and how ancillary effects analysis can be usefully integrated into policy-making processes. Finally, Section 6 outlines key steps in promoting better understanding and consideration of this important topic in policy-making.

## **2. Methodological and conceptual issues**

### **2.1 Terminology**

Ancillary benefits of GHG mitigation policies have been defined as the social welfare improvements from greenhouse gas abatement policies other than those caused by changes in greenhouse gas emissions, which incidentally arise as a consequence of mitigation policies<sup>2</sup>. This concept is not unique to climate change policy. However, the heterogeneous sources of GHG throughout the economy, their intricate economic impacts, and the global nature of climate change, make the assessment of ancillary benefits more complex than in many other policy areas. Also, due to the large uncertainties about the long-term and direct impacts of climate change, and the best methods for valuing these impacts, analysis of the shorter-term, non-greenhouse effects seems especially important if governments are to implement sensible policies in this area.

The different terms used to depict ancillary effects reflect differences in their entry into the policy process. Thus, the term *co-benefits* (sometimes also referred to as multiple benefits), signals (monetised) effects that are taken into account as an explicit (or *intentional*) part of the development of GHG mitigation policies. The term *ancillary benefits*, indicates impacts that arise *incidental* to mitigation policies. (See Figure 1a and 1b). This paper uses the term ancillary effects to denote those impacts that occur as an incidental consequence of changes in GHG emissions. This should be understood to include negative impacts, or costs, and does not imply that the ancillary effects are necessarily of lesser importance than greenhouse gas abatement. The workshop participants appreciated that distinctions between ancillary benefits and co-benefits have not always been consistently followed. There was general agreement about the need to promote finer analytic distinctions between these terms.

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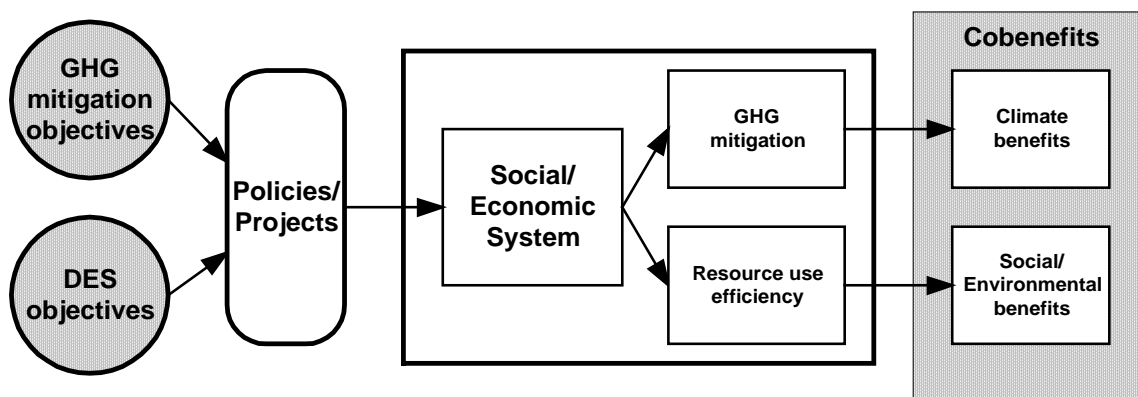
<sup>2</sup> Or, in the case of climate change adaptation policies, outcomes other than reduced vulnerability to the potential impacts of climate change - this is discussed further below.

Different scientific literatures use different terms for distinguishing physical and economic effects. The term impact in the paper always means physical effects and can be an improvement or detriment, the term ancillary benefits means the value to society of obtaining the physical improvements and the term ancillary costs means the value lost to society from negative outcomes, such as making health worse. Effect is used in this paper to be the most general term, meaning both physical and economic outcomes unless otherwise stated.

There appear to be three classes of literature regarding the costs and benefits of climate change mitigation: (1) literature that primarily looks at climate change mitigation, but that recognises there may be benefits in other areas; (2) literature that primarily focuses on other areas, such as air pollution mitigation and recognises there may be benefits in the area of climate mitigation; (3) literature that looks at the combination of policy objectives (climate change and other areas) and looks at the costs and benefits from an integrated perspective. Each of these classes of literature may have their own preferred terms.

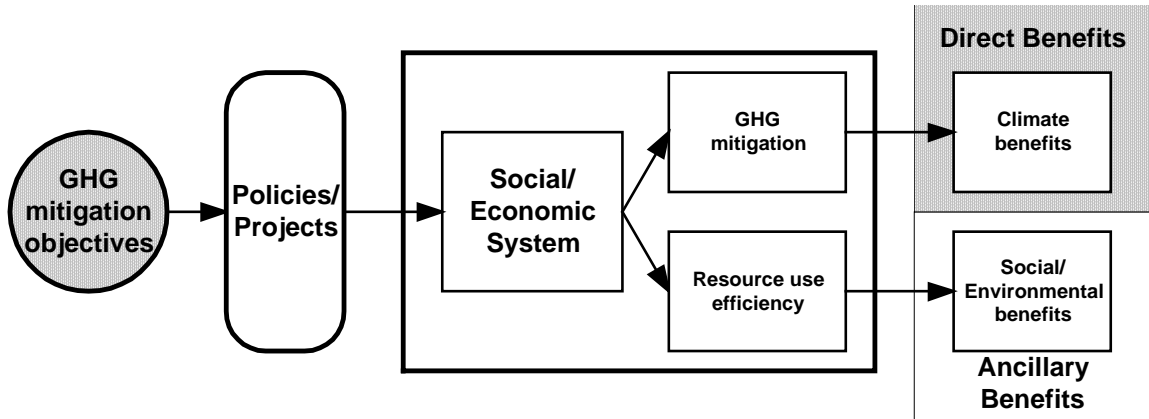
The IPCC and others are using the term “co-benefits” when speaking generically about the issues covered in class (3), in particular the integration of consideration of policies to mitigate climate change with concerns about sustainable development and other policy objectives. The terms “ancillary effects” or “ancillary benefits and costs” are used in this paper when addressing the class (1) and (2) literature. The class (1) literature appears to be the most extensive and it is this literature on *ancillary benefit and costs* of greenhouse gas emission mitigation that is primarily covered in this paper.

Figure 1a. Co-benefits of GHG mitigation<sup>3</sup>



<sup>3</sup> Figures 1a and 1b are adapted from discussions of Working Group III of the IPCC, Capetown June 2000. The term “DES” depicts development, equity and sustainability.

Figure 1b. Ancillary benefits of GHG mitigation



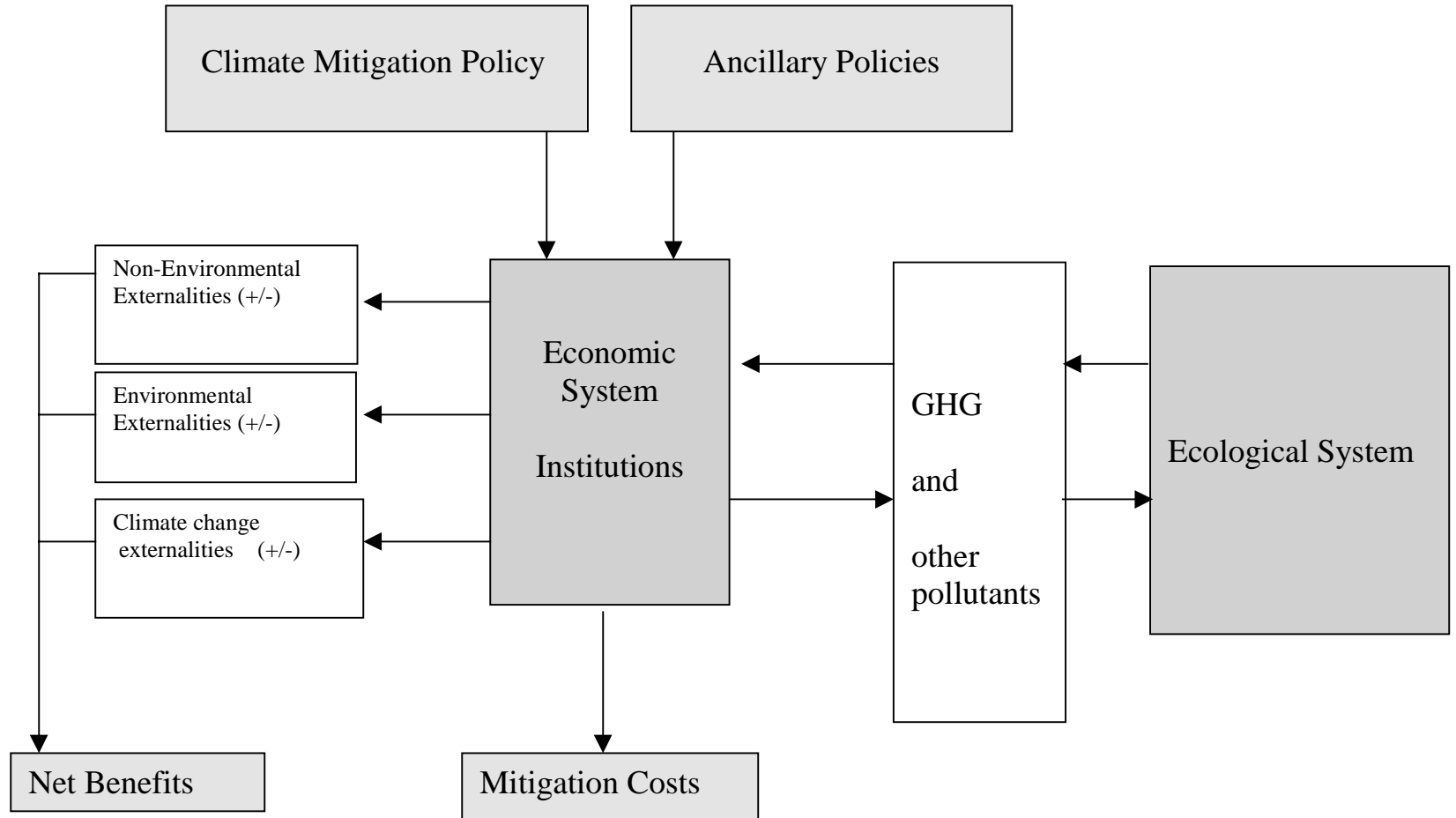
## 2.2 Conceptual framework

Whichever terminology/approach is used, Krupnick et al (2000) indicate the central importance of the economic and institutional system to determination of ancillary effects. Figure 2, provides a graphical representation of the general approach to ancillary effects analysis. Climate mitigation policies operate through an economic and institutional system within a country that leads to reductions in GHGs, changes in other pollutants, and mitigation costs. The emission changes work through an ecological or environmental system that eventually feeds back into the economic system. Then, depending on conditions of the economic system and its institutions, such as labour markets, tax systems, existing environmental and other types of regulations (represented by the box labelled “Ancillary Policies”), these feedbacks may become environmental externalities (such as changes in conventional air or water pollution), non-environmental externalities (such as employment effects) and, of course, climate change externalities (such as leakage of carbon emissions).

The importance of the economic system and institutions argues against the methodology used in early ancillary effects analyses, which implied fixed coefficients between greenhouse emissions and other effects. Different technological and regulatory structures, and differences in economic parameters, will make these relationships situation-dependent. For example, Barker and Rosendahl (2000) showed that changes in assumptions about the future price of oil can drastically change the measurement of ancillary benefits as higher prices will themselves drive many of the improvements which climate change policies might support.

Consideration of the economic system and institutions adds considerable complexity to the analysis of ancillary effects, and has implications for the types of analysis chosen. It raises questions of balancing analytical completeness with the need to limit the time and resources spent on analysis. Development of the analytic baseline - which includes projections of many of these institutional and socio-economic parameters - is clearly a vital element in analysis of ancillary effects since these baseline issues determine the environment in which climate change policies will have their effects.

Figure 2. Ancillary benefits and costs of climate change mitigation: A conceptual framework



## 2.3 *Baseline issues*

Assumptions about what will happen in the absence of any explicit policies critically determine the scope and scale of any potential ancillary effects. Morgenstern (2000) identifies and discusses five issues where baselines could be significant in assessing ancillary effects. These issues are distinct from those that are generally considered in the baselines of large-scale economic models. The first three of these issues - non-greenhouse policies, technology and economic development - are all very closely interconnected. Changes in any of these will generally have direct implications for the others. The final two - demography and natural activities - also have such linkages, but of a far smaller order, so these can usefully be treated as exogenous to GHG policy evaluation.

### 2.3.1 *Non-greenhouse policies*

Current and assumed future laws, policies, and regulations (and degree of compliance) play a major role in shaping the relevant ancillary effects baseline. As a general rule, the more abatement of ancillary effects that occurs in the baseline, the lower will be the measured ancillary effects of climate change policy. As an easy example, if it is assumed that leaded petrol will be phased out for air quality reasons, then climate change policies that reduce travel or increase vehicle efficiency will have no ancillary lead abatement benefits. On the other hand, if it is assumed that consumer preferences for environmental quality will increase over time, or the potentially exposed population will increase, then the estimated benefits of ancillary emissions reductions will be higher than if these changes had not occurred. For example, when full account is taken of the U.K.'s national target under the Long-Range Transboundary Air Pollution Convention's Second Sulfur Protocol, Burtraw and Toman (1997) estimate that the mean value of the ancillary effects calculated by Ekins (1996) for European nations declines by about one-third.

Two particular issues in relation to policy baselines include:

New regulatory activity – The pace and stringency of regulatory activity can directly affect the size of ancillary effects. More stringent regulations reduce the amount of pollution to be controlled, for instance. In addition, the precise form of new regulations can affect the size of ancillary effects. For instance, in response to a GHG mitigation policy, the hard cap on SO<sub>2</sub> emissions in the US SO<sub>2</sub> trading program results in the counting of avoided SO<sub>2</sub> abatement costs as ancillary benefits but not health improvements.

Compliance with regulations - It is not generally appropriate to assume that all emitters will be in full compliance with new or existing standards. In some developing countries where economic or other development goals take priority over environmental considerations, compliance cannot be expected to be high. Even where there is a strong history of enforcement, non-compliance can exist. For example, more than half the US population lives in areas that are currently in violation of the ambient ozone standard.

Although environmental policy is an important element of the baseline it is not the only relevant area of concern. For instance, income distribution, market reforms in energy and transport, health policies, and the location of economic activity can all have impacts on future levels of ancillary effects. At a minimum, such policy assumptions need to be made explicit.



### 2.3.2 *Technology*

While assumptions about economy-wide rates of innovation and technology/efficiency improvements are generally transparent in macro-level analyses of the costs of GHG reductions, more detailed estimates may be needed for ancillary effects analysis. Often, the effect of economy-wide assumptions on future baseline emissions is not transparent, and sometimes it is not even addressed. For instance, assumptions about the expected rate of vehicle stock turnover, fuel quality, and the decay rate of catalytic converters as the fleet ages are all critical components for estimating baseline ancillary emissions, but are not generally stated or even addressed in ancillary effect calculations.

### 2.3.3 *Economic development*

Macro-economic assumptions that are employed about baseline levels and growth rates of aggregate economic activity (GDP) will critically affect estimates of the direct benefits and costs of GHG mitigation policies. With respect to the calculation of ancillary benefits, these assumptions of large-scale factors do not generally permit specific inferences to be made about potential impacts. Disaggregation at the industry and regional level is clearly critical to understand shifts from pollution-intensive industries to the service sector. In addition, to get a full understanding of the ancillary effects it is important to understand the size of the population exposed to conventional pollution. This, in turn, requires an understanding of the spatial location of the emissions *vis à vis* the population.

### 2.3.4 *Demography*

While large-scale economic models routinely consider overall population trends, they generally do not take account of a number of other demographic factors that are important to the consideration of ancillary effects. For example, continued improvements in the health status of the population, or access to universal health care, will affect the estimation of ancillary effects in a number of ways. Increasing urbanisation tends to expand the size of the population exposed to high pollution levels. The overall ageing of the population can also affect the estimate, as the aged are more vulnerable to health damaging effects of pollution.

### 2.3.5 *Natural activities*

A final baseline issue concerns the natural resource baseline, particularly the assimilative capacity of the natural system. Many ecological processes are relatively poorly understood, but will greatly affect the calculation of ancillary effects. For example, assumptions about the time to nitrogen saturation in soils greatly affected the percentage of projected chronically acidic lakes in the Adirondacks, New York, USA. Insofar as ecological impacts are an important source of ancillary effects, better understanding of these systems is required to accurately estimate any benefits/costs.

## 2.4 *Other key issues in ancillary effects analysis*

### 2.4.1 *Developed and developing countries*

Most of the ancillary effects literature (again, here we mean effects to include physical impacts and their monetary value to society) until quite recently came from developed countries, especially the US

and Europe. Many of the data used are based on detailed, national assessments of health and other impacts and values. As examination of ancillary effects is extended into developing countries, a number of difficulties arise.

First, there is the question of which effects are direct and which are ancillary. In developed countries with quantitative commitments under the Kyoto Protocol, governments are compelled to consider alternative approaches to meeting Kyoto targets, their costs and benefits. So, there is little fundamental difficulty with the consideration of ancillary effects of climate policies in principle. However, for many countries without specific climate abatement commitments, there is a range of higher priority development and environmental concerns. In these countries, governments may be hesitant to consider health impacts related to air pollution, for example, as ancillary to climate change mitigation, since policies are far more likely to be driven by health concerns than climate change. In this instance, climate policies may not be the most effective way to address these health concerns. Raising the perspective of ancillary effects of climate policies can give a skewed view of the most efficient policies to pursue sustainable development more broadly. On the other hand, if developing countries can participate in a climate mitigation policy, through the Clean Development Mechanism (CDM) where developed countries pay for GHG mitigation in rapidly developing countries, then ancillary effects may consequently arise. Whether such steps are the most efficient from the perspective of sustainable development is less important in this context. This issue is closely related to that of baselines. If non-climate policies, such as controlling regional air pollution, are a priority for a country, then those policies should be carefully considered in estimating the baseline conditions for ancillary effects.

A second concern that arises in assessing ancillary effects in developing countries is questions about the relevance of using health and economic studies obtained in developed countries to project effects in other regions. A number of studies in developing countries employ health estimates based on work produced primarily in the U S and Europe, adjusted for GDP and sometimes other factors. It is not clear that such an approach accurately reflects differences in culture, priorities and assessment of risk. Seroa da Motta (2000), for example, shows that approaches using transfer of economic assumptions and data and those based on indigenous data provide widely diverging assessments of the value of health impacts. This study also indicates the difficulties in collecting accurate and comprehensive data in developing countries. This leaves researchers in a quandary of not being able to easily collect indigenous data, but not being confident of reliance on data transfers from developed countries. In relation to the public health impacts of various scenarios, there is a growing and fairly robust literature indicating that the scale and magnitude of physical effects is fairly well understood (Davis et al., 2000). However, it is clear that more work on appropriate data for developing countries is required before results from these nations can be accorded a high degree of reliability.

Ancillary effects should be understood and estimated in geographic and time-specific context. For many developing countries the problem is not simply ignorance about the existence of ancillary effects. Rather, decision makers have to weigh the potential ancillary effects of proposed GHG mitigation policies against other priorities. If the inclusion of ancillary effects does not tend to increase the short-term welfare of the community/society, it is unlikely that the mitigation option would be adopted. This may be the case in those developing countries where basic needs are yet to be satisfied.

#### 2.4.2 *Comprehensiveness of effects*

In order to ensure that analyses of potential ancillary effects are integrated into the policy process, it is important to consider as many types of ancillary effects as practicable. Section 3 identifies four

categories of ancillary effects: health, ecological, economic, and social. To date most research has focused on health, while limitations of both methods and data have constrained the ability to estimate the other benefit categories. More research is needed in these areas. Future work may confirm the general view that health benefits are indeed by far the most important source of ancillary benefit. However, for the moment this conclusion is chiefly a result of the fact that health impacts have been well studied and valued, in contrast to ecological, archaeological or other materials impacts, for example.

It is important to include the full array of potential impacts in the analysis of ancillary effects. For instance, omitting consideration of the environmental risks from greater reliance on nuclear or hydroelectric power, for example, could bias ancillary benefits upwards. Burtraw and Toman (1997) show that avoided costs may be an important and growing source of ancillary benefits and that is important to identify and quantify their range. For many effects, such as those relating to cultural values of historic preservation, it may never be possible to fully examine all ancillary benefits and costs.

Comprehensiveness can also affect the nature of measures taken in different sectors. For example, if only health impacts are examined when looking at transport policies, measures such as fuel efficiency or alternative fuels that affect technology but not behaviour may be favoured. But if effects on congestion and vehicular fatalities, and reduced energy efficiency, are also included, measures to alter transport behaviour may well become more attractive, even if they are not the most cost-effective measures when looking at greenhouse gas reductions or air pollution reductions alone.

Comprehensive coverage is important within classes of ancillary effects as well as between them. Often only a subset of the relevant pollutants is considered in ancillary pollutant studies. It is now widely recognised that multiple pollutants may yield significant ancillary effects. The more recent US and European studies have focused on NO<sub>x</sub>, ozone, SO<sub>2</sub>, and PM<sub>10</sub>. Given the importance of NO<sub>x</sub> for the formation of fine particle (secondary pollutants), this is a critical addition.

Of course, pollutants of interest can vary significantly by country. For example, in some developing countries where direct combustion of coal is still prevalent in the household sector, both indoor and outdoor exposures may be important. Similarly, there may be significant ancillary effects associated with reduced lead exposure in a country such as Chile, or the other nearly 100 countries where leaded gasoline continues to be used as an octane booster in gasoline (Dessus and O'Connor 2000).

#### 2.4.3 *Ancillary costs (i.e., negative ancillary effects)*

In addition to considering the full range of sources of ancillary benefits, it is also vital for analysts to consider ancillary *costs*. These can arise both from increases in externality-causing activities as well as changes in the spatial distribution of emissions. For example, while there are possibilities for increasing employment in some sectors through greenhouse abatement activities, these can also lead to a drop in employment in others. A loss of employment or income has been associated with worsened health status, including alcoholism, spouse abuse, and mental health problems (Viscusi, 1994; Perkins, 1998; Lutter and Morrall, 1994; Portney and Stavins, 1994). Consequently, the negative impacts on employment or income in some sectors may have social consequences that are not captured in economic models. On the positive side, ancillary effects should also account for any benefits arising from increases in employment or income. The point is that there is potentially a range of ancillary costs or benefits in this area.

Replacement of coal with other energy sources is often cited as a GHG abatement policy with many ancillary benefits. However, there is also potential for ancillary costs. These could come from substitution by nuclear power, for instance, which would involve health and other types of risks, by hydroelectric power with attendant externalities to river ecosystems, or by biomass from sinks based on monoculture with consequent ecological impacts. Another example would be a switch to diesel for transportation fuel, which would have a lower carbon content than gasoline but would have greater emissions of some conventional pollutants.

Also, if greenhouse abatement policies lead to substitution from electricity to more home fuel use, this could have important ancillary costs in terms of indoor air pollution particularly in developing countries where delays in electrification can also mean delays in attainment of literacy.

A further potential source of ancillary costs is the “ancillary leakage effect.” Though there is debate about the significance of the effect, it is widely observed in modelling the impacts of Annex I actions to reduce GHG emissions that carbon emissions in non-Annex I countries may rise, due to changes in relative factor prices. The resulting increase in coal use (and in use of other fossil fuels) in non-Annex I countries—the carbon leakage—brings with it an ancillary cost of greater air pollution and other negative externalities. Because control efficiencies of conventional pollutants are lower in developing countries than in developed countries, and, perhaps, population densities near power plants and other large users of energy may be larger in developing countries, ancillary costs may be larger than suggested by carbon leakage or fuel use changes (Wiener, 1995). Preliminary analysis of extant modelling results suggests the possibility of ancillary costs resulting from increases in conventional pollutants in developing country regions as a consequence of “leakage effect” of carbon reductions under the Kyoto Protocol (Krupnick, Burtraw, and Markandya, 2000) However, the issue is not well-studied and the significance of the effect is not known.

Finally, Lutter and Shogren (1999) point out that ancillary costs could arise from the geographical reallocation of economic activity following a carbon mitigation policy. If carbon trading were in place, for instance, some areas, relative to their carbon allocation baseline, would be net sellers, others net buyers. In extreme cases, some net buyers could actually exceed their BAU carbon and conventional pollutant levels. Such cases may be far fetched. However, the possibility exists that net carbon permit buyers have facilities in or near dense, urban areas, while net sellers do not. In this case, net population exposures to ancillary pollutants could increase, , even with constant aggregate carbon emissions.

It is interesting to note that the examples of ancillary costs given above relate to ‘macroeconomic’ policy options rather than ‘micro’ decisions where specific investment decisions consider technologies to limit or eliminate greenhouse gas emissions. Although ancillary costs could also arise at this ‘micro’ decision level, they are less likely to be as significant. This underscores the point that the kinds of ancillary costs and benefits considered depend on the policies and technologies being evaluated, local and regional demographic characteristics, and their specific national and institutional context.

#### *2.4.4 Alternatives to economic valuation*

Much of the controversy around ancillary effects really concerns the issue of valuation, especially how risks to human health and loss of life are valued (see Davis, Krupnick, and Thurston 2000 in this volume). This issue is one which has been the subject of considerable discussion in the literature for many years (Grubb, 1999). However, there is no inherent need for ancillary effects analysis to engage in valuation per se. Rather, this analytic decision is one of trade-offs. On the one hand, the valuation of ancillary impacts conceptually takes place according to public preferences for the different types of

impacts. Many would think that this approach is better than having decision-makers substitute their own preferences for the public's. Such valuation permits social benefits to be compared to social costs of mitigation. On the other hand, valuation is highly controversial, with much uncertainty that is not always reflected in valuation analysis. Decision-makers routinely take actions that weigh economic and other impacts, including health impacts, against each other. Attempting to value these may sometimes obscure, rather than make more transparent, the decisions that are being made. This is both in terms of final decisions, and in engaging other players in the decision-making process.

An example of a relevant study outside the benefit-cost framework is a study of options to achieve greenhouse and air quality benefits simultaneously in four case study areas by STAPPA/ALAPCO (1999)<sup>4</sup>. Some of the main results are presented in Table 1. The STAPPA/ALAPCO case studies focused on the potential greenhouse and conventional pollutant reductions that could occur in four U.S. sample areas if harmonised strategies, defined as those strategies that simultaneously reduce conventional pollutants and greenhouse pollutants, were implemented. The areas differed in emissions, economic and energy profiles, making the reductions only broadly comparable. Here, no valuation is used, and air quality decision-makers can readily see the implications for emissions of climate change policy options. Similarly, some analysts and decision-makers may be inherently more comfortable with analysis which deals with human health and mortality impacts without valuing in monetary terms (see, for example, Working Group on Fossil Fuels, 1997, which estimated that the range of avoidable deaths that could occur globally by 2020 under some GHG mitigation policies extended from 4 to 11 million.)

**Table 1. Percent Reduction from Baseline Emissions in Four Case Study Areas, due to implementation of a package of climate change abatement measures**

Area	SO <sub>2</sub>	NO <sub>x</sub>	PM	VOC	CO	CO <sub>2</sub>
New Hampshire	41%	17%	12%	3%	4%	12%
Atlanta, GA	40%	6%	1%	3%	4%	7%
Louisville, KY	26%	14%	3%	3%	4%	15%
Ventura County, CA	2%	4%	1%	4%	4%	11%

Source: STAPPA/ALAPCO 1999.

While of some value to decision makers, these types of estimates are not easily compared without a framework for assessing their economic impact and the efficiency with which various proposed targets and GHG reductions can be achieved. Participants at the workshop raised the option of performing cost-effectiveness analysis of alternative policies instead of cost-benefit analysis as a way to improve policy-making while avoiding the controversies and uncertainties of valuation.

#### 2.4.5 Location of polluting activity

This is most obviously important in the case of air pollution. The social costing literature has vividly demonstrated that the benefits of emission reductions can vary tremendously depending on the spatial location of emission reductions vis-a-vis the proximity of the exposed population. Krupnick and Burtraw (1997) and earlier studies reconciling U.S. and European estimates for the social costs of fuel

<sup>4</sup> The executive summary of the STAPPA/ALAPCO study is also reproduced in this volume.

cycles found that population density differences between Europe and the U.S. account for 2 to 3 times larger benefit estimates in Europe. Meteorology and other factors, including the potential for a non-linear relationship between emissions and pollutant concentrations, or between concentrations and health effects, further enhance the value of complex, location-specific models.

Pollution crosses over boundaries separating economies and societies that have different institutions, wealth and culture. This is particularly an issue in Europe, where transboundary pollution is an important element of regional policy making. ExternE work (EC 1995) pointed out that the externalities of energy use would be greater if it were not assumed that all pollution stopped travelling as soon as it reached the EU border. One of the explanations given for the lower estimated values of ancillary effects in the US relative to Europe is that more US pollution falls into the ocean where it has no health impacts. In fact, some of this pollution travels to Canada and estimates would be higher if these transboundary effects were included.

Ancillary impacts, such as changes in noise and ecosystem impacts also vary geographically. Even issues such as employment impacts will vary depending on the location of the effect.

#### 2.4.6 *Uncertainty*

There is general agreement that the uncertainty surrounding the estimates of ancillary impacts is at least as great relative to the value of those estimates as that associated with other mitigation costs. The process by which external costs and benefits are calculated involve a number of physical modelling steps and a valuation step. The modelling involves estimation of emissions, their dispersion and transformation, and the impacts of the pollutants. The valuation of the impacts is based on statistical techniques that also have large error bounds. Each of the steps also has some uncertainty associated with it in terms of modelling choices. And the cumulative uncertainty, which is a combination of model and statistical uncertainty, could be quite large.

A good study of ancillary costs and benefits will provide some idea of how large the statistical uncertainty bounds are. A single number is indicative of a misleading approach and of less than thorough analysis. There is more than one way to report the uncertainty. For the statistical uncertainties, it is possible to derive probability intervals, using Monte Carlo methods, or by other statistical methods. For model uncertainty, other methods such as bounding analysis, breakeven analysis or meta analyses have been used. Finally a method that integrates both types of uncertainty based on subjective and objective error estimates is that of Rabl and Spadaro (1998). This method provides a quantification of the uncertainty and, recognising that many studies do not have enough information to carry out a quantitative analysis, reports a subjective qualitative indicator of uncertainty. For climate change work, Rabl and Spadaro (forthcoming) suggest that model uncertainty be described as follows:

- “Well Established”: models incorporate known processes; observations consistent with the models; multiple lines of evidence support the cost assessment.
- “Well posed debate”: different model representations account for different aspects of observation/evidence, or incorporate different aspects of key processes, leading to different answers. Large bodies of evidence support a number of competing explanations.
- “Fair”: models incorporate most known processes, although some parameterisations may not be tested representations; observations are somewhat inconsistent and incomplete.

Current empirical estimates are well founded, but the possibility of changes in governing processes is considerable. Possibly only a few lines of evidence support the evaluations.

- “Speculative”: conceptually plausible ideas that have not received much attention in the literature or that are laced with difficult to reduce uncertainties.

At the least, ancillary benefit studies should provide similar qualitative information about uncertainty. In doing so, however, it is important not to overstate uncertainties, or to let “the perfect be the enemy of the good.” Policy analysis of any importance always deals with considerable uncertainty, and judgements must be made as to the value of analytic resources relative to value of more certainty.

### **2.5 Use of modelling for ancillary effects analysis**

Because of the underlying complexities of specific industry and geographic factors, disaggregated models represent a superior approach for developing accurate estimates of ancillary effects (again, including the monetization of physical effects). Aggregate models, which have many advantages for the study of GHG mitigation policies, are not well suited to capture the important detail or non-linearities involved in estimating ancillary effects.

There has generally been a lack of interface between large scale economic modellers and ancillary effects experts. The clear advantage of large-scale economic models is their ability to incorporate general equilibrium effects not available in the simpler models. In contrast, the disaggregated models have the capacity to generate geographic-specific results.

Debate over the appropriateness of large-scale versus disaggregated models has been an issue in climate change policy-making for many years. While there has been progress in bringing the two approaches together analytically, comprehensive ancillary effects analysis appears to require more detailed disaggregation. Therefore, in looking at methodologies to improve analysis of ancillary effects, attention to the development of models will be required. As well as more aggregate detail, these models will need to explicitly handle a range of emissions and environmental impacts, alternative approaches to environmental (and other) policies, and model the linkages between climate change policies, other policies, and economic and institutional factors (including technological change).

### **3. Categories of ancillary effects**

Ancillary effects are most commonly thought of as “direct” changes in outcomes, most commonly health, ecological, economic/welfare, and, perhaps, congestion, and safety. However, in some cases, ancillary effects will be in the form of avoided costs, where the actual outcomes are the same, but the costs of achieving these outcomes is reduced. In terms of the types of climate policies examined, these have been almost exclusively *abatement* policies. However, it is almost inevitable that some level of climate change will already occur due to anthropogenic interference in the climate. Therefore, it is also worthwhile to consider ancillary effects of policies to *adapt* to climate change. Each of these issues is discussed briefly below.

### 3.1 Health

Most efforts to estimate ancillary effects of mitigation policies have focused on avoided deaths and illness tied with exposure to particulate matter in developed countries. Recent work indicates that there is a broader array of important air pollutants and associated health impacts, not all of which have been quantified at this time. Borja-Aburto et al, 2000 , provide meta-analyses on some of this recent work, finding increased mortality and morbidity associated with ozone and particulate matter. Table 2 indicates health effects that have been quantified, along with those that are not usually incorporated into such quantifications.

Table 2. Scope of health effects

<b>Human Health Effects of Air Pollution</b>	
<b>Quantifiable Health Effects</b>	<b>Non-quantified/Suspected Health Effects</b>
Mortality*	Neonatal and post-neonatal mortality
Bronchitis - chronic and acute	Neonatal and post-neonatal morbidity
Asthma attacks	New asthma cases
Respiratory hospital admissions	Fetus/child developmental effects
Cardiovascular hospital admissions	Non-bronchitis chronic respiratory diseases
Emergency room visits for asthma	Cancer (e.g., lung)
Lower respiratory illness	Behavioral effects (e.g., learning disabilities)
Upper respiratory illness	Neurological disorders
Shortness of breath	Respiratory cell damage
Respiratory symptoms	Decreased time to onset of angina
Minor restricted activity days	Morphological changes in the lung
All restricted activity days	Altered host defense mechanisms
Days of work loss	(e.g., increased susceptibility to respiratory infection)
Moderate or worse asthma status	Increased airway responsiveness to stimuli
	Exacerbation of allergies

Source: Adapted from: The Benefits and Costs of the Clean Air Act 1990 to 2010, U.S. EPA, EPA-410-R-99-001 1999).

The Workshop also considered that important interactions take place between poverty (or income, more generally), nutrition and pollutant exposure (Davis et al., 2000). Among the factors that may increase susceptibility to air pollution are:

1. enhanced susceptibility to pollution for populations with existing compromised health status, due to genetic predisposition, impaired nutritional status, or severity of underlying disease);
2. greater per capita exposures to atmospheric pollution in the center of cities than in the general population, due to greater pollutant density combined with a lower access to protective environments, such as air conditioning;
3. exposures to various residential risk co-factors such as indoor cooking fuels, rodents, cockroaches, dust mites, and other indoor pollution sources (e.g., gas stoves used for space heating purposes); and/or
4. increased prevalence of poverty, which is associated with reduced access to routine preventive health care, medication, and health insurance.



Most analyses of the ancillary effects of mitigation policies have looked at health effects associated with reductions in criteria (conventional) pollutants from energy combustion, including avoided deaths, acute and chronic illnesses, such as bronchitis, respiratory diseases and asthma, and behavioural effects, such as restricted activity days. There are other potentially important areas of health impacts, including occupational health and safety risks associated with, e.g. coal mining, forestry and the nuclear fuel cycle. In some cases, these risks may be wholly or partially internalised so that the ancillary effects of removing them may be less than expected.

Health effects typically account for 70-90% of the total value of ancillary benefits (Aunan *et al* , 2000, this volume) and so deserve special attention in ancillary effects analysis. The dominance of health impacts in ancillary effects analysis can qualitatively alter the analysis. Without better estimates of other impacts besides health, archaeological and ecological effects will generally not be relevant to decision-making on GHG mitigation.

The distribution of health effects between regions and among the population may differ. In places where the unemployment rate is high, the amount of Willingness To Pay (WTP)/Willingness To Accept (WTA) for avoiding such health impacts may be lower or the estimate of losses of earning due to illness may be lower<sup>5</sup>. The poor section of the population may suffer more than the rich as they have to spend higher proportion of their income on medical care. Such distributional impacts can be of great significance in assessing the ancillary health effects.

There are many complexities in valuing health impacts, and a very extensive literature on this topic, which is not reviewed here. It is notable, however, that the US EPA, in reviewing the evidence on this topic, identified a plausible range of \$1.6 million - \$8 million for the value of a statistical life, with a central estimate of \$4.8 million (US EPA, 1999). This range is large, even when looking only at one country. In extending analyses across countries, further uncertainties are introduced. Davis, Krupnick, and Thurston (this volume) discuss the sensitivity of ancillary benefit estimates to assumptions about the mortality risk coefficient and the Value of a Statistical Life (VSL) Routine values used in the literature can lead to a difference of 300% in ancillary benefit estimates.

Given the importance of health effects to overall ancillary effects analysis, this level of uncertainty is important to how information is assessed. Three potential approaches to dealing with the prevalence and uncertainty of health impacts are:

- sensitivity analysis around plausible ranges. This is generally considered an important element of any analysis of complex issues;
- use of conservative estimates - for example relying on estimates which are the minimum acceptable estimates, perhaps based on direct health costs. While this approach should avoid arguments over the minimum level of ancillary effects, it may not be very helpful in determining optimal policy choices;
- avoiding valuation of health impacts - it may be that decision-makers are more comfortable with comparing health impacts directly with other impacts, including financial impacts, so that this approach could avoid considerable uncertainty while still assisting policy choice. However, this approach makes comparative and comprehensive analysis difficult and rules out normative policy analysis.

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<sup>5</sup> These terms are explained in the Appendix to Krupnick, Burtraw, and Markandya in this volume.

The state of the science of valuation of health effects is currently in ferment, with serious questions being raised about the inappropriateness of basing the valuation of mortality risks of the type affected by air pollution on labor market studies. *Ad hoc* adjustments for the shorter life span of those thought to be most affected by air pollution (the elderly and ill) have been made, but more credible estimates of the willingness to pay to avoid such impacts awaits new research. In developed countries, such efforts are more likely to lower such estimates relative to current estimates than raise them (see Davis, Krupnick, and Thurston, 2000, for a full discussion).

### 3.2 Ecological

Many experts believe that ancillary ecological benefits, though largely unstudied, may well be an important category of ancillary effects. Rothman (2000) indicates some of the areas where greenhouse policies could have significant ecological impacts (see Table 3). Climate change policy analysis often assesses land based abatement/sink policies as being more cost-effective than, for example, energy sector policies. This could be enhanced if there were significant positive ancillary effects of greenhouse policies in the energy sector. On the other hand, if there are significant negative ancillary effects, they could alter this analysis and lead to significant shifts in perceptions of relative costs of sectoral policies.

Table 3. Policies and pressures

GHG Policy	Pressure on Ecosystems				
	Harvesting	Waste residuals	Physical restructuring	Magnified Extreme Events	Exotic Species Introductions
Curtailment of Energy Use		+			
Changes in Energy Extraction and Production Methods		+	++		
Improvements in Energy Efficiency		++			
Fuel Switching	+	++	++		
GHG Capture		+	+		
Increase or Maintain the Area of Land in Forests		+	++		+
Manage Forests to Store More Carbon	+	+	++	+	+
Manage Non-Forested Lands to Store More Carbon	+	+	++	+	+
Reduce Dependence on Fossil Fuels Through Product Substitution	+		+		

+: potentially small effects; ++: potentially large effects

Source: Rothman 2000.

Krupnick *et al* (1998) finds that airborne NO<sub>x</sub> emissions slated to occur under the 1990 Clean Air Act significantly reduce nitrate loadings in the Chesapeake Bay. Aunan *et al.* (1998) suggests that forests in large parts of Europe are probably adversely affected by air pollution although, as they note, “the understanding of the causes and mechanisms is poor except in the most polluted areas where direct effects are plausible.” It is thus reasonable to assume that ecological ancillary benefits will arise from reductions in airborne emissions, although these have not yet been specifically modelled. A modelling effort recently established in Europe is beginning to look beyond airborne emissions and focus on direct water discharges associated with GHG policies (RIVM *et al.* 2000). Various types of both user and non-user benefits are likely to be tied with both air and water pollution, although, as indicated, they have not yet been specifically modelled as ancillary effects of GHG reduction policies.

Lack of available studies on ecological ancillary effects and land use impacts is an important gap in the knowledge base, as is indicated in the report from the Workshop regarding data gaps and research priorities.<sup>6</sup>

### **3.3 Other**

The most commonly cited source of other ancillary effects are safety and congestion, both of which are especially important in transport. In the same paper in which he examined economic/welfare ancillary effects, Barker (1993) found that even a small tax increase would lead to a significant reduction in fatal and non-fatal road accidents. Sommer has recently extended this work (2000)<sup>7</sup> in several European countries, finding that the annual toll from air pollution associated with vehicles is equal to that linked with road injuries. Outside of transport, safety impacts could arise from shifts in the nature of production (e.g. shifting from coal mining to solar cell production) although the marginal impacts due to policies could be small and hard to measure. In such cases, it would be important to consider effects across all affected sectors, not just those where the safety impacts are in one direction.

Proost (2000) cites congestion as the overriding ancillary effect of transport in developed countries, outweighing even health impacts. However, there is some question as to whether these effects are internalised already to transport users in the aggregate. If already internalised, ancillary effects would *not* be counted. Pearce (2000) also cites community severance as an ancillary impact related to transport, where roads divide ecosystems and social systems with consequent ecological and quality-of-life impacts.

Aunan et al. (2000), projected significant reductions in materials damage from implementation of energy efficiency programs in Hungary, and suggested significant increases in crop yields were likely to be obtained if NO<sub>x</sub> and VOC emissions were reduced in large regions in Europe.

### **3.4 Equity**

Social equity among different socio-economic groups remains of paramount concern to policy makers grappling with climate change. Concerns over relative regional impacts are also important, for example when governments consider impacts of carbon taxes on regional industries. But equity can also include a concept that different sectors of the economy should each bear a “burden” broadly equivalent to the share of emissions they cause, an effect noted by Bonney (2000). All of these equity impacts are fundamentally different from other ancillary effects. They relate to the distribution rather than the total share of costs and benefits. While this is clearly vital to policy-makers, it would broaden the scope of ancillary effects analysis far beyond what is manageable to include it. Equity issues warrant specific consideration in policy-making.

### **3.5 Economic**

Economic ancillary effects can include a diverse range of issues. In some cases, it is questionable whether these effects may justifiably be labelled as ancillary effects of climate policies, and it is especially important here to distinguish primary and ancillary effects. For example, the energy cost

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<sup>6</sup> For more information on this see [www.wri.org](http://www.wri.org).

<sup>7</sup> In this volume.

savings that derive from a fuel efficiency policy are probably best seen as a direct financial benefit that should be offset against the costs of the policy. The most commonly discussed categories of economic effects are:

*Ancillary financial impacts* - These are financial impacts that are easily quantifiable, but do not derive directly from the policy put in place. They are often hard to distinguish from direct costs and benefits of a policy. So, with the energy efficiency example above, while fuel savings are best seen as a direct cost saving of the policy, changes to maintenance costs may be seen as an ancillary effect or may be simply added to the fuel savings costs. In the end, the key is to ensure all of these effects are included in the analysis somewhere. Examples include projected economic benefits of 0.05% of 1990 GDP from ancillary effects including road surface maintenance expenditures associated with implementation of an EU carbon tax (Barker 1993).<sup>8</sup> Such impacts can be very specific to the policy chosen, and so can be resource intensive to analyse and compare among policy options, especially as the ability to quantitatively examine these effects will vary.

*Employment change* - Climate policies clearly have potential to create or reduce jobs in a sector or geographic region. However, it is unclear whether these should generally be seen as ancillary effects. This is first because such impacts must really be looked at on an economy-wide basis - and in principle this should include an examination of the job impacts of raising money if the policy involves raising government funds. Also, in a fully employed economy, economic analysis indicates that much of the job impacts will be temporary, but there are still transitional costs. So, employment-related ancillary impacts are difficult to estimate as they require general equilibrium analysis at the same time as detailed sectoral and/or geographic analysis. Further, one must be wary of double-counting employment-related impacts and direct or general equilibrium costs. For this reason, most cost-benefit analyses include employment impacts under a discussion of distributional effects. As a minimum, inclusion of these effects should be detailed and transparent. On a pragmatic basis, it is noted that potential employment *losses* are routinely considered in policy-making, although not always in a transparent way, and not in the context of the flow-on effects of employment changes discussed above (under “ancillary costs”).

*Energy security* - Guaranteeing reliable, affordable energy has been an important objective of national governments since at least the first oil shocks, although the relative importance has declined in recent years. Most of the justification for concerns about energy security has stemmed from events outside the normal operation of markets, including cartel behaviour and war, making standard economic analysis difficult. ExternE work on this topic indicates it is likely to be of small magnitude relative to other effects. (European Commission 1998)

*Induced technological change.* Depending on policies proposed, induced technological change may or may not be an example of an ancillary effect. The important principle is consistency - if a policy redirects technological innovation, the losses as well as the gains must be included. There is a small but growing literature specifically focused on induced technological change, i.e., how much additional economy-wide innovation, if any, can be stimulated by GHG mitigation policies. However, there is no strong consensus of views in this evolving field (see, for example, Grubb et al, 1995; Goulder and Schneider, 1999; and Goulder and Mathai, 1998). If GHG mitigation

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<sup>8</sup> There have been a number of studies indicating that, through appropriate recycling of carbon tax revenue, GDP can be increased through such a mechanism. Such results are the subject of considerable discussion in the literature and it is debatable whether these should be included as side effects of climate policies. From a pragmatic perspective, such impacts are *relatively* well-understood and evaluated and so do not need specific attention in the ancillary benefits framework.

policies do accelerate the overall rate of technological change, then this would feed through to increases in GDP and competitiveness.

### **3.6      *Avoided costs***

Avoided costs refer to the cost savings that come from achieving a given outcome by introducing a climate change policy. So, the final health, ecological, or social outcomes are the same, but at a lower cost. The most commonly cited example is cost savings for meeting the SO<sub>2</sub> cap in the United States, as examined by Burtraw and Toman (1997), US EPA (1999), and Burtraw *et al* (1999). If the SO<sub>2</sub> cap is binding, moderate policies to reduce GHG emissions from the power and industrial sectors will not lead to further reductions of SO<sub>2</sub> emissions. Inasmuch as these emissions are capped, the result is abatement cost savings to those purchasing or otherwise acquiring SO<sub>2</sub> permits freed up by the GHG policy's induced SO<sub>2</sub> reductions. Burtraw *et al* (1999) estimate that these avoided costs could be equivalent to the direct ancillary benefits of a moderate greenhouse tax, therefore doubling the estimate of ancillary benefits (although this study also finds that such estimates are less than mitigation costs, significantly so for high carbon tax regimes). Avoided costs may also arise in other regulatory regimes, where companies have choices as to how to achieve a given environmental performance. The importance of avoided costs relative to direct ancillary effects is likely to grow over time, where reliance on cap and trade programs is expanded, or where negotiated settlements about how to meet ambient targets are used. Many previous ancillary effects analyses have failed to take into account these possible developments.

### **3.7      *Adaptation***

The IPCC Third Assessment Report distinguishes between responses to direct consequences of climate change, which are referred to as adaptation, and efforts to reduce or prevent these effects, which are termed mitigation. The Workshop did not concentrate on the former issue, beyond noting that adaptation to the effects of climate change generally receives less attention than mitigation in most countries' policy making processes. There is no denying that historical emissions have likely already made some climate change inevitable. Taking action to address vulnerability to these changes could provide ancillary benefits and costs.

For example, decisions to build sea walls or wildlife corridors as adaptation measures could lead to wider ecological positive or negative impacts. Preparations for the increased spread of disease could encourage improved general or specific medical care. Measures taken to increase irrigation efficiency in preparation for reduced availability of freshwater due to seawater intrusion could also benefit agriculture and increase water for hydropower and drinking (see Scheraga 1999).

Given the relative scarcity of specific adaptation measures undertaken by countries, it is probably not worthwhile at this stage to consider ancillary effects analysis for these measures specifically. However, in developing any general approaches, it will be important to consider potential secondary or ancillary effects of adaptation policies.

## 4. Evidence from case studies

### 4.1 Ancillary public health benefits from GHG mitigation and comparison to mitigation costs

To assist with a systematic assessment of the impact on public health from GHG mitigation, Table 4 summarizes studies that have devised methods for estimating and valuing health impacts, including some studies presented at the Workshop. The table outlines the regions and scenarios assessed, the pollutant pathways and endpoints considered, , as well as the resulting estimates of ancillary benefits in 1996\$ per ton carbon. Table 5 shows the modelling methods employed in these studies and some basic characteristics of these assessments.

Burtraw and Toman (2000) (this volume), Kverndokk and Rosendahl (2000), and Ekins (1996) have all recently reviewed ancillary benefit studies finding that ancillary benefits can be from 30% to over 100% as large as gross (i.e., private) mitigation costs. For all of these studies, the benefits should be viewed as “very crude,” because of use of simplistic tools and transfers of dose-response and valuation functions from studies done in other countries. For instance, some studies rely on expert judgement instead of established dose-response functions and estimates of national damages per ton rather than distinguishing where emissions changes occur and exposures are reduced. In these circumstances, large differences in ancillary effects per ton across several Norwegian studies can be attributed to differences in energy demand and energy substitution elasticities. If carbon-based energy production is reduced rather than switched to less carbon-intensive fuels, ancillary effects will be far larger. However, some studies did not consider the “bounceback” effect when a less carbon-intensive technology is substituted for a more intensive one in response to a carbon mitigation policy.

Kverndokk and Rosendahl (2000) have also assessed ancillary benefit studies that feed environmental benefits back into the economic model, and find that this modeling difference can significantly enhance estimated ancillary benefits. Recent work from the International Co-Benefits Program of the U.S. EPA, in conjunction with the governments of Chile, Brazil and China have produced some particularly useful results. They indicate that the adoption of readily available energy efficiency technologies in transportation, industry, and residential uses can provide a scale of ancillary effects equal to the costs of adoption of those GHG mitigating policies (see for example, Cifuentes et al., 2000).<sup>9</sup>

#### 4.1.1 Summarizing the ancillary benefit estimates

The broad divergence in the value of ancillary effects estimates, even within the same country is evident in Table 4. Across countries, values range from around \$2 to more than \$500 per tonne of carbon abated, with the lowest estimates in the US and the highest in Chile and Norway. Where studies include uncertainty bounds, these are often quite large relative to the central estimate.

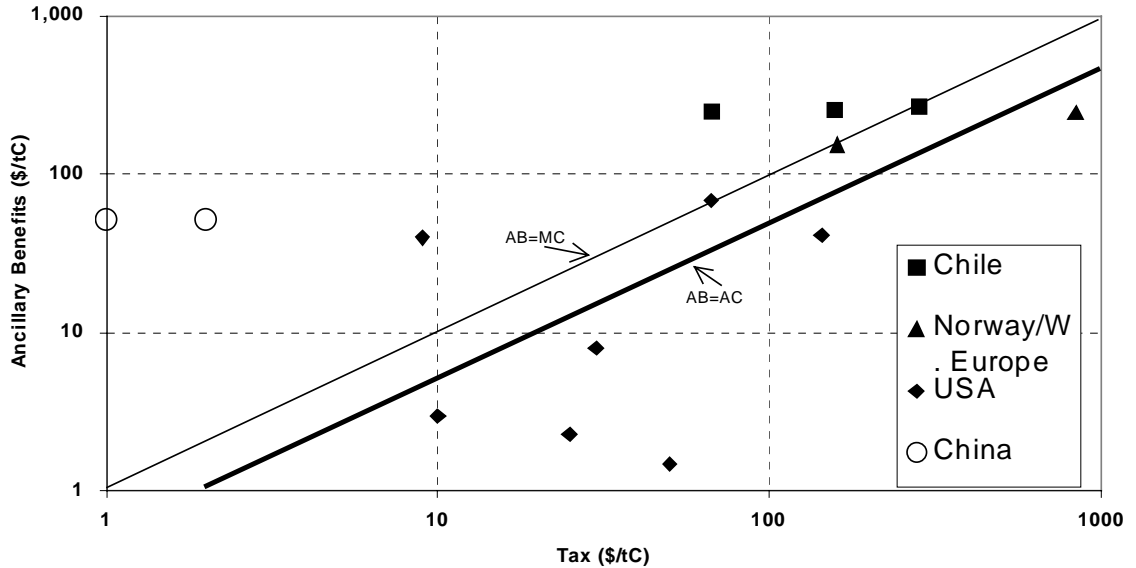
Figure 3 displays estimated ancillary effects per ton relative to the size of the carbon tax imposed (in \$1996/tC). Points on the diagonal line  $AB=MC$  indicate equality between the two measures (because marginal costs (MC) will equal the tax rate in theory). Some points are on this line; more appear above it than below, with the studies on Norway and western Europe and the U.S. split. If abatement costs are assumed to be a square function of emission reductions, average costs can be computed as one half of marginal costs, with the corresponding diagonal line  $AB=AC$ . As more points appear

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<sup>9</sup> Cifuentes et. al. 2000 present results for Chile. Preliminary result from Brazil and Korea were also presented at the workshop - for more information see the website on the workshop: <http://www.oecd.org/env/cc/>

above the line than below, this indicates that ancillary benefits could be equal to or even exceed the costs of mitigation in some instances.

Figure 3. Ancillary effects in 1996US\$/tonC versus levels of the carbon tax



Source: authors.

As for the change in ancillary benefits per ton C with a change in carbon taxes, there are differences in results. Burtraw et al (1999) show this ratio falling dramatically in percentage terms with higher carbon taxes<sup>10</sup>, while Dessus and O'Connor (1999) show it rising slightly and Abt (1999) shows it rising dramatically. The Abt result arises because they assume that the proposed U.S. SO<sub>2</sub> cap becomes non-binding considerably below the higher tax rate modelled. In addition, the National Ambient Air Quality Standards are treated as a cap by Abt, with reductions in pollution below these “caps” treated as benefits but reductions above these caps treated as saving abatement costs.

It is not surprising that estimates of the size and scale of ancillary effects could and should diverge. This is because of differences in policy scenarios, modelling and parameters. In addition, there are real differences across countries, such as population size, regulatory differences, technological sophistication and baseline emissions of conventional pollutants. However, without a consistent methodological base against which to assess these matters, it is impossible to determine which of the differences in study results derive from “real” differences and which derive from alternative methods. Examining these studies against some of the issues identified in preceding sections provides some clarification of these differences and of the relative role played by various components.

<sup>10</sup> Although it should be noted that this does not include consideration of avoided costs, which may be expected to rise relative to other types of ancillary benefits.

Table 4. Scenarios and results of studies reviewed

Study	Area and Sectors	Scenarios (1996 US\$)	Average Side effects \$/tC (1996US\$)	Key Pollutants	Major Endpoints
Dessus and O'Connor, 1999	Chile (benefits in Santiago only)	1. Tax of \$67 (10% C reduction) 2. Tax of \$157 (20%) 3. Tax of \$284 (30%)	1. \$251 2. \$254 3. \$267	7 air pollutants	Health – morbidity and mortality, IQ (from lead reduction)
Cifuentes, et. al. 2000	Santiago, Chile	Energy efficiency	\$62	SO <sub>2</sub> , NO <sub>x</sub> , CO, NMHC Indirect estimations for PM <sub>10</sub> and resuspended dust	Health
Garbaccio, Ho, and Jorgenson, 2000	China – 29 sectors (4 energy)	1. Tax of \$1/tC 2. Tax of \$2/tC	1. \$52 2. \$52	PM <sub>10</sub> , SO <sub>2</sub>	Health
Wang and Smith, 1999	China – power and household sectors	1. Supply-side energy efficiency improvement, 2. Least-cost per unit global-warming-reduction fuel substitution, 3. Least-cost per unit human-air-pollution-exposure-reduction fuel substitution		PM, SO <sub>2</sub>	Health
Aunan, Aaheim, Seip, 2000	Hungary	Energy Conservation Program	\$508	TSP, SO <sub>2</sub> , NO <sub>x</sub> , CO, VOC, CO <sub>2</sub> , CH <sub>4</sub> , N <sub>2</sub> O, VOC	Health effects; materials damage; vegetation damage
Brendemoen and Vennemo, 1994	Norway	Tax \$840/tC	\$246	SO <sub>2</sub> , NO <sub>x</sub> , CO, VOC, CO <sub>2</sub> , CH <sub>4</sub> , N <sub>2</sub> O, Particulates	Indirect: Health costs; lost recreational value from lakes and forests; corrosion  Direct: Traffic noise, road maintenance, congestion, accidents



Table 4 continued

<b>Study</b>	<b>Area and Sectors</b>	<b>Scenarios (1996 US\$)</b>	<b>Average Side effects \$/tC (1996US\$)</b>	<b>Key Pollutants</b>	<b>Major Endpoints</b>
Barker and Rosendahl, 2000	Western Europe (19 regions)	Tax \$161/tC	\$153	SO <sub>2</sub> , NO <sub>x</sub> , PM <sub>10</sub>	Human and animal health and welfare, materials, buildings and other physical capital, vegetation
Scheraga and Leary, 1993	US	\$144/tC	\$41	TSP, PM <sub>10</sub> , SO <sub>x</sub> , NO <sub>x</sub> , CO, VOC, CO <sub>2</sub> , Pb	Health – morbidity and mortality
Boyd, Krutilla, Viscusi, 1995	US	\$9/tC	\$40	Pb, PM, SO <sub>x</sub> , SO <sub>4</sub> , O <sub>3</sub>	Health, visibility
Abt, 1999	US	1. Tax \$30/tC 2. Tax \$67	1. 8 2. \$68	Criteria pollutants	Health – mortality and illness; Visibility and household soiling (materials damage)
Burtraw et al., 1999	US	1. Tax \$10/tC 2. Tax \$25 3. Tax \$50	1. \$3 2. \$2 3. \$2	SO <sub>2</sub> , NO <sub>x</sub>	Health

#### 4.1.2 Evaluation of the studies

Almost all the studies of ancillary effects reviewed here analyse the effects of a GHG reduction policy through a tax on carbon. The ranges of the tax extend from modest levels (9 yuan/tC in 2010 for Garbaccio et al. (2000), \$10/tC for Burtraw et al (1999)) to high levels (254 \$/tC Dessus and O'Connor (1999), \$840/tC for Brendomoen (1994)). The US studies employ relatively modest taxes, between \$10/tC up to \$67/tC. Only two studies consider alternative programmes: Aunan 2000 considers a National Efficiency Programme, and Cifuentes et.al. 2000 considers energy efficiency improvements, based on the adoption of existing technologies. The level of abatement of these two studies is relatively modest. How do the different studies compare in terms of the issues identified above?

##### Baseline issues

**Other policies** – One of the key differences in study approaches relates to assumptions about baseline regulatory policies. For instance, Burtraw et al (1999) and Abt (1999) count the abatement cost savings from reducing SO<sub>2</sub> emissions in response to a carbon tax because SO<sub>2</sub> emissions are capped in the U.S. Similar adjustments are not made for SO<sub>2</sub> taxation (or taxation of other pollutants) in Europe, where large differences exist in regulatory policy.

**Economic development** - A major reason for differences in findings relates to whether the values for health impacts are increased with future income growth. Several of the developing country studies follow this approach. In general, developed country studies do not. This may create significant inconsistencies in comparisons of ancillary effects across countries. The U.S. Science Advisory Board has endorsed the idea of adjusting for economic growth. However, there is significant uncertainty concerning income elasticity of the willingness to pay for anticipated health improvements. A number of studies have found elasticities in the 0.2-0.6 range based on income differentials *within* a country. Such elasticities, when applied to transfers *among* countries, yield higher values than the default elasticity (1.0) used by most of the developing country-studies reported in Table 5.

**Demography** - None of the potential impacts of changing demographic profiles is explicitly incorporated into the ancillary effects literature, with the exception of Burtraw et al. (1999) which included population projections according to geography, age and income in their analysis.

**Comprehensiveness of coverage** – The major focus of all the studies is on changes in mortality associated with projected changes in particulate exposure, with differential handling of this issues (see below). Many studies did not go beyond particulate associated mortality, although a few included morbidity and there was a wide scattering of other issues covered in the other studies. Most studies did not include consideration of avoided costs, although Burtraw et al (1999) and Abt (1999) consider avoided costs due to the SO<sub>2</sub> cap.

Most of the studies rely on concentration-response functions from the health literature, and apply them using a standard methodology (Ostro 1996, US EPA 1999). The most important health effects are premature mortality and chronic respiratory effects.

Table 5. Modeling choices of studies reviewed

Study	Baseline (as of 2010)	Economic Modeling	Air Pollution Modeling	Valuation	Uncertainty treatment
Dessus and O'Connor, 1999	4.5%/yr economic growth; AEEI: 1% Energy consumption: 3.6% PM10: 1% Pb: 4.1% CO <sub>2</sub> : 4.8%	Dynamic CGE	Assumed proportionality between emissions and ambient concentrations	Benefits transfer used: PPP of 80% U.S. VSL: \$2.1 mil VCB: \$0.2 mil IQ loss: \$2500/point	Sensitivity tests on WTP and energy substitution elasticities
Cifuentes et al, 2000	For AP control, considers implementation of Santiago Decontamination Plan (1998-2011).	No economic modeling. Only measures with private, non-positive costs, considered	Two models for changes in PM <sub>2.5</sub> concentrations. 1) Box model, which relates SO <sub>2</sub> and CO <sub>2</sub> to PM <sub>2.5</sub> 2) Simple model assumes proportionality between PM <sub>2.5</sub> concentrations apportioned to dust, SO <sub>2</sub> , NO <sub>x</sub> and primary PM emissions.; Models derived with Santiago-specific data and applied to nation	Benefits transfer from US values, using ratio of income/capita Uses original value for mortality decreased by 1 std. dev. VLS = \$407k in 2000	Parameter uncertainty through Monte Carlo simulation. Reports center value and 95% CI
Garbaccio, Ho, and Jorgenson, 2000	1995-2040 5.9% annual GDP growth rate; carbon doubles in 15 years; PM10 grows at a bit more than 1% per year,	Dynamic CGE model; 29 sectors; Trend to U.S. energy/consumption patterns; Labor perfectly mobile; Reduce other taxes; 2-tier economy explicit	Emissions/Concentration coefficients from Lvovsky and Hughs; 3 stack heights	Valuation coefficients from Lvovsky and Hughs; VSL: \$3.6 mil (1995) to 82,700 Yuan in 2010 (income elas =1). 5%/ year increase in VCB to \$72,000	Sensitivity analysis
Wang and Smith, 1999		No economic modeling	Gaussian plume	Benefit transfer using PPP. VSL=\$123.700, 1/24 of US value	

Table 5 continued

Study	Baseline (as of 2010)	Economic Modeling	Air Pollution Modeling	Valuation	Uncertainty treatment
Aunan, Asbjorn and Seip, 2000	Assumes status quo emissions scenario.	Two analyses: Bottom up approach and macroeconomic modeling	Assumed proportionality between emissions and concentrations	Benefit transfer of US and European values using 'relative income' = wage ratios of 0.16	Explicit consideration through Monte Carlo simulation. Reports center value and (low, high) (at which CL?)
Brendemoen and Vennemo, 1994	2025 rather than 2010. 2%/year economic growth, 1% increase in energy prices, 1-1.5% increase in electricity and fuel demand CO <sub>2</sub> grows 1.2% until yr 2000, and 2% thereafter	Dynamic CGE		Health costs of studies reviewed based on expert panel recommendations. Contingent valuation used for recreational values	Assume independent and uniform distributions
Barker and Rosendahl, 2000	SO <sub>2</sub> , NO <sub>x</sub> , PM <sub>10</sub> expected to fall by about 71%, 46%, 11%) from 1994-2010	E3ME Econometric model for Europe		\$/emissions coefficients by country from EXTERNE: 1,500 Euro/t NO <sub>x</sub> for ozone; NO <sub>x</sub> and SO <sub>2</sub> coefficients are about equivalent, ranging from about 2,000 E. to 16,000 E. per ton; PM <sub>10</sub> effects are larger (2,000-25,000). Uses VSLY rather than VSL: 100,000 E (1990).	
Scheraga and Leary, 1993	1990-2010 7% growth rate C Range for criteria pollutants 1-7%/year	Dynamic CGE			

Table 5 continued

Study	Baseline (as of 2010)	Economic Modeling	Air Pollution Modeling	Valuation	Uncertainty treatment
Boyd, Krutilla, Viscusi, 1995		Static CGE		\$/emissions coefficients	
ABT, 1990	2010 baseline scenarios – 2010 CAA baseline emission database for all sectors. Plus at least partial attainment of the new NAAQS is assumed. Benefits include getting closer to attainment of these standards for areas that wouldn't reach them otherwise. Includes NO <sub>x</sub> SIP call	Static CGE		From Criteria Air Pollutant Modeling System (used in USEPA RIA and elsewhere)	SO <sub>2</sub> sensitivity – SO <sub>2</sub> emissions may not go beyond Title IV requirements;  NO <sub>x</sub> sensitivity – NO <sub>x</sub> SIP Call reductions not included in final SIP call rule
Burtraw et al, 1999	Incorporates SO <sub>2</sub> trading and NO <sub>x</sub> SIP call in baseline;	Dynamic regionally-specific electricity sector simulation model with transmission constraints. The model calculates market equilibrium by season and time of day for three customer classes at the regional level, with power trading between regions.	NO <sub>x</sub> and SO <sub>2</sub> . Account for conversion of NO <sub>x</sub> to nitrate particulates	Tracking and Analysis Framework: The numbers used to value these effects are similar to those used in recent Regulatory Impact Analysis by the USEPA.	Monte Carlo simulation for CRF and valuation stages.

All studies in Table 5 account for the best studied pollutant in the public health pathways—particulates. Most, however, do not consider secondary particulate formation from SO<sub>2</sub> and NO<sub>x</sub>, or do so in a very simplistic manner. None of the studies of ancillary effects considered ozone-related morbidity or mortality. In a developed country, direct particulate emissions are likely to be a large fraction of particulate mass, making the lack of attention of secondary products less important. In developing countries, however, secondary products are likely to be far more important than primary particulates, and ozone can be quite important, especially in some meteorological zones. Omitting these products could bias ancillary benefit estimates downwards; using proportionality assumptions or other simple approaches raises uncertainties and may carry biases. Only one study considers lead emissions; few address ozone, which is widely acknowledged to increase morbidity, with much more uncertainty about its effect on mortality. The Abt study (1999) is the most comprehensive in its modelling of secondary particulate formation and dispersion, finding that 12 urban areas in the U.S. would come into compliance with the recently promulgated standard for PM<sub>2.5</sub> (*which has been remanded by the court and is not yet in effect*), for a carbon tax of \$67 (\$1996); otherwise these areas would not be able to meet the new standard. With there being little information on PM<sub>2.5</sub> concentrations in the U.S. urban areas; these estimates should be viewed as highly speculative.

Besides the differences of the base rate of the effects reflecting underlying age distribution, other factors account for the different outcomes of the studies. First, some studies use PM<sub>10</sub>, while others use fine particles (PM<sub>2.5</sub>), or even some components of them (sulphates and nitrates). When the individual components of PM<sub>2.5</sub> are used, their risk is assumed to be similar to that of PM<sub>2.5</sub>. To date, this has not been verified (especially for nitrates, the secondary particulate product from NO<sub>x</sub> emissions). Second, studies that look at age groups separately generally report higher impacts. Aunan et al (2000), for example, used a much steeper dose-response coefficient for people older than 65 yrs than used by other studies. Third, different studies consider different endpoints. This is especially important for mortality estimates. Most of the studies consider only associations with daily deaths, obtained from time-series studies. Very few (Abt, 1999, is one) consider the chronic effects on mortality, derived from cohort studies (e.g. Pope et al, 1995). Use of the latter effects can produce estimates of deaths that are three times larger than use of the former. Also, only a few studies consider effects on child mortality or morbidity.

A number of studies consider transportation-related consequences of a carbon tax. There are many significant issues in converting such changes to externalities, which are not addressed here.

None of the studies reviewed in this assessment reported estimates of ancillary costs.

**Location** – The level of spatial detail varies very widely, from the fine detail of Burtraw et al (1999) to national level evaluations including the international summation of national figures in Barker and Rosendahl (2000). Many studies extrapolate data from a single region or site to much broader coverage, while Dessus and O'Connor (1999) limit their analysis to the Santiago region. With the exception of the European assessments, none of the studies considers transboundary issues of ancillary impacts outside the study area.

**Treatment of Uncertainty.** Several of the studies use Monte Carlo simulation and other, less sophisticated techniques for characterising uncertainties. In addition, many conduct sensitivity analyses on key economic, health, and valuation parameters to estimate the range of possible ancillary effects.

**Economic Modeling.** Most of the studies in Table 5 use static or dynamic CGE models. One employs an econometric model which provides top-down and sectorally aggregate estimates of ancillary effects/costs (Barker et al, 2000). The modelling of carbon reductions as a result of a policy

intervention, such as a tax, is credible though subject to key choices about energy substitution and demand elasticities. Although restricted to the electricity sector, the Burtraw et al (1999) model provides the sole example of location-specificity of an economic model. Because of its restricted focus, this analysis can provide more credible modelling of population exposure reductions than that generated from spatially aggregate models. Its detailed representation of investment choices along with the endogeneity of these choices also distinguishes this study and the model behind it. Finally, several studies do not use an economic model, but follow a bottom up approach, positing some increase in energy efficiency or reduction in carbon and estimating the ancillary effects that would result, at a reasonably detailed spatial level. Such studies suffer from not accounting for behavioral adjustments, such as energy substitutions, that could alter their estimates of ancillary effects considerably. The high ratio of ancillary effects to the carbon tax for Garbaccio, Ho, and Jorgenson (1999) assessment of China appears to be due to very optimistic assumptions about these elasticities.

#### 4.1.3 *Why studies for the same country differ*

Clearly, studies can differ substantially in their treatment of many of the conceptual issues discussed above. Consider why the estimates of ancillary effects (costs) from two different studies of Chile differ. Dessus and O'Connor (1999) estimate benefits of about \$250/tC where as Cifuentes et al (1999) estimate benefits of about \$62/tC. Half of the Dessus and O'Connor benefits are tied with effects on IQ due to reduced lead exposure, an endpoint not considered by Cifuentes et al. The large lead-IQ effect is not consistent with US and European studies on this neurotoxic conventional air pollutant, but could in part be due to the relatively high exposures that currently occur in Chile.

Also, the VSL used by Dessus and O'Connor is more than twice as large as that used by Cifuentes (\$2.1 million vs. \$0.78 million). These choices were driven by alternative benefit transfer approaches. Dessus and O'Connor used 1992 (purchasing power parity) to transfer a US VSL, while Cifuentes et al used 1995 per capita income differences and the exchange rate. This comparison points out the importance of the choice of benefit transfer approach in estimating ancillary effects.

These differences aside, it appears that other modeling choices, which appear to be very different across the two studies, had little effect on the results. For instance, Dessus and O'Connor used a top-down model, while Cifuentes used a bottom-up approach.

For the US, Abt (1999) finds for a carbon tax of \$30, ancillary effects per ton are \$8. This includes mortality and morbidity. Burtraw et al (1999) find that for a \$25 carbon tax, the ancillary effects per ton are \$2.30. If avoided cost benefits of \$3 are added,<sup>11</sup> the difference in costs is not that large. For a \$50 per ton tax, Burtraw et.al. find ancillary effects of only \$1.50/tC, while for a slightly larger tax (\$67), Abt estimates that ancillary effects are \$68/tC. Why the large disparities here?

First, the Burtraw et al analysis uses mortality potency factors for NO<sub>x</sub> (i.e., particulate nitrates) that are about one-third of those used by Abt and the factors used to value mortality risk reductions are about 35% lower in Burtraw et al (who adjust the VSL for the effects of pollution on older people rather than on averaged aged people). Second, Burtraw et al study is restricted to the electricity sector and is highly disaggregated and dynamic. The restriction to the electricity sector results in lower public health benefits than to the entire economy because, by 2010, NO<sub>x</sub> emissions per unit carbon are projected to be lower for this sector than in the general US economy. Third, Abt finds that there are significant ancillary cost savings, i.e. the \$67 carbon tax is large enough to bring SO<sub>2</sub> emissions significantly under an SO<sub>2</sub> cap that is 60% lower than the current cap, and it brings NO<sub>x</sub> emissions

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<sup>11</sup> With a \$10 carbon tax, Burtraw et.al. (1999) find \$3/tC in ancillary benefits.

down low enough to bring significant numbers of non-attainment areas into attainment with the national ambient standards. It is unclear whether a \$67 carbon tax would be large enough to promote such reductions.

In addition, Burtraw et al do not account for new, tighter ozone and PM standards being implemented in the U.S., but Abt does (while assuming only partial attainment of the standards). This baseline assumption should result in lower emissions of conventional pollutants to be controlled in the Abt study than in the Burtraw et al study and would in principle bring down the Abt estimates of ancillary benefits in comparison.

## **5. Impacts on policy making processes**

The assessment of potential ancillary effects can influence choices about the stringency and types of GHG mitigation policies that may be adopted. Depending upon the local, national and regional priorities, the understanding of potential ancillary effects can play a major role in affecting policy tools (e.g. taxes versus regulation versus voluntary agreements) as well as the sectoral, technological and geographic focus. Despite this importance, ancillary effects have not generally received systematic treatment in policy-making.

### **5.1 *Ancillary effects and the policy “toolkit”***

Climate change mitigation policy designed to comply with the Kyoto Protocol is still being developed in Annex I countries but the signs are that there will be a mix of economic, regulatory, voluntary and information instruments (OECD 1999b). There is a limited literature which considers how ancillary effects analysis affects the choice of policy instrument, with Pearce (2000) and Krupnick, Burtraw and Markandya (2000) providing some of the first comprehensive examinations of this issue.

Economic instruments (such as carbon taxes or tradable carbon quotas) have clear economic advantages over other environmental policies, and incorporating ancillary benefits/costs into such instruments is conceptually straightforward. By calculating the benefits, the level of a tax can be raised, or the allocation of quotas lowered to account for them.<sup>12</sup> However, incorporation of ancillary benefits alters a key advantage of economic instruments – the ability to allow abatement to take place wherever it is most cost-effective. Greenhouse gases affect the global climate in the same way regardless of their geographic source. Ancillary effects are, however, more localised, so that the location of GHG abatement affects the overall benefit achieved from a policy. Including ancillary effects in greenhouse policy design could mean geographically targeting GHG control, or spatially-differential taxation. In the case of emission trading regimes, it could lead to localised restrictions on carbon trades. So, consideration of ancillary effects can complicate “simple” economic instruments for GHG abatement at the national, regional or international level. It could, for example, affect the perceived optimal balance between domestic abatement and participation in international flexibility mechanisms.

Some regulatory approaches may lend themselves more readily to incorporation of ancillary effects. For example, standards based on Best Available Technology (BAT) could define ‘best’ technology as that which achieves not only some defined carbon emission target but also other associated targets. These targets could include many ecological and safety effects. Incorporation of ancillary effects will be more difficult in moving from BAT to ‘practicable’ or ‘reasonable’ technology standards, since

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<sup>12</sup> Or, in the case where there are net ancillary costs, the tax lowered or the quota raised.



increasing the range of incorporated effects will tend to make standards more environmentally stringent than such approaches would tend to support. Technology-based standards may also lead to higher costs in areas such as employment and induced innovation, as they can be more expensive than alternative approaches and/or redirect technology innovation away from more cost-effective opportunities.

“Voluntary” agreements between polluters and government are increasingly used in environmental policy (OECD 2000). These agreements can take a range of forms, but in general appear easily capable of incorporating ancillary effects, as the agreements are very flexible as to what parties wish to include in them. This can be done in ways that reflect very specific local circumstances, although increasing specificity will lead to increasing complexity and probably time for negotiation. As a minimum, agreements can ensure that approaches to one environmental problem do not lead to ancillary costs in relation to another.

Education/information programs seem superficially able to incorporate ancillary effects. Consumers can be presented with information about a range of impacts from product or service choices. However, for these programs to really incorporate ancillary effects, there would need to be some synthesis of information on the range of impacts. For example, adding a “greenhouse efficiency” rating to a dishwasher that already had ratings for energy efficiency, water consumption and recyclability, would add information but in a manner which required consumers to integrate across these pieces of information. This again would be similar in concept to adding a carbon tax to a range of environmental taxes. A single greenhouse rating adjusted for other impacts would be required for full incorporation of ancillary effects, and such an approach seems unlikely. Information programmes based on more qualitative information may be better suited to incorporation of ancillary effects.

Overall, inclusion of ancillary effects is likely to make *design* of GHG abatement policies more complex, especially in adding a geographic dimension which need not otherwise exist. In terms of *selecting* among policy instruments, different instruments do differ in their ability to incorporate ancillary effects. However, all instruments appear capable of building in ancillary effects to some degree. Other selection criteria of general economic efficiency, environmental effectiveness and equity, remain central to instrument selection.

## 5.2 *Ancillary effects and sectoral policies*

Inclusion of ancillary effects can affect not just the type of policy instrument put in place, but the sectoral targets of policies. Where policies are aimed at specific technologies, they may also affect the choice of technological options.

One obvious example is diesel fuel for transport. As a greenhouse measure, substitution of diesel for petrol can be a very cost-effective greenhouse abatement opportunity. However, when ancillary health impacts are included, the serious health costs of diesel make it look less attractive than other options (Pearce 2000).

However, in transport, the overall handling of ancillary effects is not so clear. Broadly, transport sector measures can be divided into those which reduce the overall level of traffic, and those which reduce emissions from a given level of traffic (such as alternative fuels or fuel efficiency). If health effects from air pollution are the overriding ancillary effect in the transport sector, then the two types of measures are both reasonably attractive. However, if, as Proost (2000) suggests, congestion is the overriding ancillary effect (at least in peak hours), then measures which reduce traffic will have far greater ancillary effects than those which reduce the emissions intensity of traffic. If this is the case,

traffic-reducing measures can be preferred to others, even if they may appear less cost-effective from a greenhouse-only perspective. And such measures can be “no regrets” measures even when the costs are relatively high relative to the greenhouse abatement effect.

At a cross-sectoral level, it is commonly felt that measures such as reforestation or afforestation and land use change can be highly cost-effective greenhouse abatement options relative to measures taken in the energy sector. However, with evidence that 60% or more of the cost of energy sector measures can be offset by ancillary benefits, the relative cost-benefit assessment can change dramatically. In the case where ancillary benefits outweigh abatement costs, as found in some studies, the relative cost-assessment could be completely switched around unless there are comparable ancillary costs in non-energy sectors.

### **5.3 *Ancillary effects and the policy making process***

Clearly, governments sometimes take ancillary effects into account in policy-making related to climate change. One example is the decision by a number of countries not to allow any (more) nuclear power stations, and in some cases to support early shutdown of nuclear power stations, despite the fact that this may make achievement of greenhouse objectives more difficult. In these cases, the environmental costs of nuclear power are assessed as being greater than the potential greenhouse gas abatement benefits. Examination of support programs for diesel fuel is another common example. What is not clear is whether most policy processes have a *systematic* approach to consideration of ancillary effects.

#### **5.3.1 *Cost-benefit analysis***

When well conducted, cost-benefit analysis (CBA) automatically accounts for ancillary benefits and costs, as it accounts for all “with” and “without” outcomes. So, more uniform use of this technique could be an important step toward extending policy analyses of potential GHG mitigation efforts. CBA is widely used for regulatory impact appraisal in the US, the European Commission now regularly subjects planned Directives to cost-benefit appraisals, and there is strong support for cost-benefit in the UK and Scandinavia. Other countries are known to experiment with cost-benefit analysis, but most decision-making is only partially informed by quantitative techniques generally, whether cost-benefit or some other technique.

There are a number of potential barriers to the wider use of cost-benefit analysis to incorporate ancillary effects in climate change decision-making, including:

- Complexities of the analysis, and disagreement over issues such as the valuation of human health impacts, make the CBA technique sometimes controversial. Given the potential importance of these valuations to ancillary impacts overall, differences in valuation techniques and results can lead to confusion for policy-makers.
- CBA is technically complex, involving appropriate selection of discounting rate, time horizon and the assessment and comparison of results; it requires considerable resources and skills, which may not always be in abundant supply.
- Cost-benefit analysis (and other formal guidance procedures) may be seen as a limit on policy-makers because it does not account for political conflicts (see, for example, EFTEC 1998).

- Institutional structures of decision-making within government militate against fully integrated policy making. For example, decision-makers on climate change are often at different levels of government, or in different Ministries, from those making decisions about local or regional air quality. Even within Ministries, there is often lack of co-ordination between policy-makers examining different issues.
- Health information is usually employed in CBA only when sufficient numbers of studies have been conducted on humans. This effectively makes proof of human harm the basis for analysis. More sophisticated use of experimental information and modelling simulations with respect to potential health impacts could reduce this problem.

So, while it is worthwhile to pursue wider use of CBA, in practice there will be limits on its use, thus it is important to consider other techniques that allow for consideration of ancillary effects in practice. CBA can never be fully comprehensive and in some cases may be very partial. In doing so, and considering the difficulties raised above, it is possible to divide alternative approaches into analytic issues (the first two points above) and institutional issues (the last two points).

### 5.3.2 *Simpler analytic structures*

The most important step in ensuring consideration of ancillary effects analysis is to ensure that the major sources of ancillary effects are identified. This paper identifies a classification of ancillary effects that could provide the basis for a checklist of effects. With more detailed consideration of case studies in a number of countries, more detailed checklists could be developed. These could be further developed through computer packages that identify likely ancillary effects of particular types of policies. A further refinement would be to include information, based on existing studies, of the likely magnitude of such benefits, and what factors affect the likely magnitude. In areas such as air pollution emissions and health impacts, there is considerable information in OECD countries, which might be used to provide such a database. Such a database could have sufficient geographical diversity to allow for more detailed consideration of likely effects. However, in other areas such as ecological impacts, there would be very little existing information to draw on. Information outside OECD countries is also very sketchy.

Use of such checklists, even at a basic level, will help to ensure more systematic consideration of ancillary effects issues. A further step in complexity is to seek to quantify impacts in commensurate terms other than money. Multi-criteria analysis allows this to happen without formally having to monetise effects such as human health. However, it is not clear that using metrics other than money avoids any of the problems of monetisation, since weighting of different categories of impacts is still required. A hybrid form of analysis, in which some of the less controversial impacts are monetised and then compared with other major impacts, may provide a sufficient degree of analytic rigour and simplicity while allowing final decision-makers flexibility to consider tradeoffs. An effective dialogue between decision-makers and analysts would help the effectiveness of such an approach.

An alternative (or possibly complementary) approach is to develop improved analytic tools to handle ancillary effects analysis. This paper points out the weaknesses of many “top-down” models in including ancillary effects in the analysis of climate change policies. Work on improving the spatial detail of such models, and more specific handling of ancillary impacts, would greatly assist policy makers. Improved models could have endogenous links between climate policies, other policies, technological change and economic development (per Figure 2). However, the complexity of such approaches should not be underestimated and the ability of models to make great progress in this area in the short term will be limited.

### 5.3.3 *Institutional approaches*

Institutions for government decision-making are generally not conducive to the conduct of ancillary effects analysis. The tendency for Ministries to focus on a single set of issues, and even within Ministries for issues to be compartmentalised makes the consideration of impacts other than primary effects difficult. So, it is likely that institutional reform will be required, although not in drastic ways. Many governments are already taking steps in this direction. Given the comprehensiveness of greenhouse gas emissions throughout the economy, ancillary effects analysis offers an analytic construct which can support efforts toward more integrated decision-making.

An important element will be the provision of tools to assist analysts. As noted above, this could involve models, but could also be simple checklists that identify the most likely sources of ancillary effects, together with a mandate for their use to ensure at least qualitative treatment of ancillary effects. In developing a new Canadian action plan on greenhouse abatement, many sectoral committees were asked to identify the ancillary effects of potential measures, while a central “roll-up” group was given the task of integrating these different effects into a co-ordinated economic analysis. While the result was not consistent across all sectors, this central direction did result in somewhat more consistent approaches to examining ancillary impacts than previously<sup>13</sup>. More detailed guidance on possible ancillary effects could be helpful in eliciting more systematic attention to this issue.

Institutional steps to support ancillary effects analysis are similar to those required for environmental impact analysis (EIA). However, a key difference will be the need to allow for EIA of policies rather than of projects and the analysis may be required at an earlier stage in the decision-making process. Links between policy-EIA and ancillary effects analysis may be useful in developing common institutional approaches.

## 6. **Conclusions and further steps**

A number of studies have estimated the hypothetical ancillary effects of future climate change policies applied in particular countries or regions. Some studies find that the benefits of the health effects avoided by mitigation measures, per ton of carbon, are roughly equal to the carbon tax/ton needed to meet those goals or even exceed the tax. Others find relatively small ancillary benefits. Thus, it is difficult to generate broad, general estimates of the magnitude of ancillary effects relative to mitigation costs. The spread of results is due to methodological differences in these studies, gaps in the models and data used in these estimations, as well as “real” differences in economies and other factors across countries. However, there appears to be compelling evidence that ancillary benefits may be a significant fraction of or even larger than the mitigation costs, especially where baseline conditions involve relatively high levels of pollution and there are likely to be minor ancillary *costs*. This is true even in developed countries, where baseline conditions include long-standing regulatory programs and lower levels of pollution.

With respect to baseline considerations, most of the literature on ancillary effects fails to systematically consider future government policies and regulations with respect to environmental policies. Other regulatory policy baseline issues, such as those relating to energy, transportation, and health, have been generally ignored, as have baseline issues that are not regulatory, such as those tied to technology, demography, and the natural resource base (Morgenstern, 2000). Adoption of more stringent regulatory regimes will result in significant reductions in potential size and scale of ancillary

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<sup>13</sup> The effects, if any, on final policy decisions, remains to be seen.

benefits. Where such regimes are not implemented, the potential for ancillary benefits can remain quite high.

The models most in use for ancillary benefit estimation – mostly CGE models – do a reasonable job estimating carbon reductions and mitigation costs from various policy interventions. But they also have the most difficulty in estimating such ancillary effects because they rarely have the necessary spatial, demographic or technological detail.

The studies reviewed here report that the biggest share of the ancillary effects is related to public health, but it is recognised that there are significant types of ancillary benefits or costs that have not been quantified or monetised, or even studied very carefully. The valuation of human health impacts is uncertain and crucial to determination of the relative importance of health and other ancillary effects. While there is a reasonable literature on this subject in developed countries, the developing country database is small.

In relation to policy choice, most studies are focused on the question of how ancillary effects analysis might affect the optimal level of policy response. Relatively little work has been done on how this issue affects the choice of specific policy tools or sectors for greenhouse gas abatement.

Further research would help to develop better understanding of ancillary benefits and costs, their magnitude and implications, especially in developing countries. At the same time, enough information is available to indicate that countries should include consideration of ancillary benefits and costs in their policy development if they are to promote cost-effective, integrated climate change policies.

In relation to further research, high priority areas for further research include:

- more targeted case studies on non-health ancillary effects, especially ecological impacts, some of which are related to air pollution;
- more comprehensive generation and use of health information on morbidity and mortality tied with the array of air pollutants of interest;
- more studies on the willingness to pay to reduce health risks, particularly in developing countries;
- more sophisticated assessments of baseline health and social conditions as these influence susceptibility to pollution in various regions;
- transparent and reasonable specification of regulatory baselines, particularly with respect to future air pollution regulation;
- development of indigenous data in developing countries, at least enough to assess how accurate benefit transfers from developed countries really are;
- development of integrated modelling, which allows simultaneous consideration of macro-scale and geographically specific impacts;
- better modelling to incorporate avoided costs, and integrated achievement of multiple policy goals;
- analyses that attempt to capture ancillary costs; and

- consideration of the time frame over which ancillary effects are realised: and the relationship to GHG policy timeframes.

In relation to incorporation in policy-making, the potential magnitude and impact of ancillary effects argues that ignoring them in climate policy making, especially at this relatively early stage can lead to important and costly errors. These could affect the level of response (including the balance between domestic response and international flexibility), the types of policy instrument used and the sectoral targeting of policies. There is no denying that the many uncertainties in ancillary effects analysis require a careful and transparent approach, including consideration of ancillary costs, if major policy mistakes are to be avoided. A “retreat to safe borders,” where only the most certain information is included, could avoid these potential mistakes, but miss out on important insights. To avoid these problems, analytic transparency and better information is required.

In the short term, the methodological framework and summary of types of ancillary effects presented in this paper and in the proceedings volume provides a skeleton upon which countries can build their ancillary effects analysis. In many cases, specific data will be lacking, but reference to previous studies, with appropriate allowance for differences in methodologies and situations, will provide some indication of likely magnitudes of ancillary effects. An implicit assumption in economics is that it is better to make tradeoffs using public preferences (expressed in monetary units) than to use preferences of the decision-makers. While comparisons of non-economic impacts can result in loss of analytic rigour, they can make the analysis of ancillary effects less controversial and this can be of value to the policy process. Assessments of lives potentially saved or hospitalisations avoided, for example, can sometimes provide a more understandable context against which policy makers may consider their options, and can avoid the additional uncertainty from the valuation step. Many of the complexities involved in ancillary effects analysis are not subject to analytic resolution, nor are they likely to be resolved within most governments’ resources.

Institutional reform will be especially important, including with respect to ensuring that critical information is gathered and monitored. Case studies of how different governments integrate ancillary effects into climate policy-making could be of benefit in identifying and assessing successful approaches. Centralised directives about appropriate methods and information to be gathered to ensure that ancillary effects are included in relevant policy-making will provide an important first step. In the longer-term, development of simplified, quantifiable methodologies for assessment will be required, especially with respect to impacts on land use, ecosystems, materials damage, archaeological resources, and other potential ancillary effects, for which there are no uniform methods of assessment available at this time.

In developing countries, there is far more uncertainty. The fact that these countries are not members of Annex I and have no current quantitative commitments under the Kyoto Protocol, allows more time for developing understanding of ancillary effects before policies are put in place. However, even here ancillary effects analysis can help countries who are potential recipients of CDM projects to assess which types of projects might lead to the greatest overall sustainable development benefits (WRI 1999).

The Workshop papers and the wider literature make clear that ancillary effects are potentially significant and warrant consideration in climate change policy-making. To date these effects are generally handled in an ad hoc, incomplete and/or inconsistent manner. The complexities involved are not to be underestimated, and there is scope for considerable further research and development to develop better methods and data for the systematic estimation of ancillary effects. This would help to ensure that ancillary effects are better integrated into policy development which in turn would improve greenhouse mitigation policies.

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**I. FRAMEWORK FOR ESTIMATING ANCILLARY  
BENEFITS AND COST**

# THE ANCILLARY BENEFITS AND COSTS OF CLIMATE CHANGE MITIGATION: A CONCEPTUAL FRAMEWORK

by Alan KRUPNICK, Dallas BURTRAW and Anil MARKANDYA

## 1. Introduction

Within the broad set of climate change issues, one that is growing in controversy and potential importance is the ancillary benefits and costs of policies to reduce greenhouse gases (GHGs). Ancillary benefits and costs are externalities arising from GHG abatement policies that are achieved jointly with the reduction of GHGs in the atmosphere.<sup>14</sup> Analysts have attempted to identify and in some cases to estimate these benefits, with estimates ranging from a small fraction of GHG mitigation costs to largely offsetting them.<sup>15</sup>

In this paper, we demonstrate and emphasize that such a variation in estimates should be expected, and indeed is a requisite criterion for the studies to be considered credible as a group. But, definitions, underlying assumptions, modeling procedures and parameters have all varied, rendering comparisons and analysis unnecessarily difficult, and under mining the quality of the studies considered individually. We articulate a principle to improve the quality of this literature: in different applications, *methods should be consistent and estimates should not*. The field needs an overarching conceptual framework, to provide guidance on how to achieve methodological consistency across studies and to help achieve a consensus with regard to how the results can be interpreted, and incorporated in climate policy discussions. Our paper begins to build this framework.

A great deal is at stake in how this controversy is decided. If these ancillary benefits are significant, or the distribution of ancillary benefits is expected to be spatially concentrated, then perhaps the development and implementation of climate policy should be altered. At the very least, knowing that the possibly high cost of climate change mitigation might be largely offset by ancillary benefits could speed up and spread the commitment to action as well as implementation itself. On the other hand, if these effects are “small” relative to the other costs or the benefits of reducing GHGs, perhaps they can be safely ignored in the debate over climate change mitigation policy — at least from the perspective of efficiency — simplifying an already too complex debate.

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<sup>14</sup> Nomenclature for this issue is still evolving. We use ancillary benefits and costs to refer to monetized effects of GHG policies. “Co-benefits” is another term in the literature. We reserve this term for the benefits of policies, say for the reduction of conventional air pollutants, that are coordinated with a climate change mitigation policy. “Ancillary impacts” refer to ancillary physical effects. Unless otherwise made clear in the text, “ancillary benefits” also refers to “negative benefits, i.e., “ancillary costs.”

<sup>15</sup> See Burtraw *et al.* (1999) and reviews by Pearce (2000); Burtraw and Toman (1997); Ekins (1996).

Although there are a number of difficult issues that have to be addressed, fortunately, the issues surrounding analysis of ancillary benefits are *not* the same order of complexity as issues surrounding the analysis of climate change generally. It is widely acknowledged that climate change is a phenomenon that transcends generations and cultures, with dramatically differing potential impacts on various peoples and countries. This, coupled with a tremendous heterogeneity in the sources of emissions and the wealth of nations, all of which are expected to change over time, raises what are perhaps the most complicated issues in economics and philosophy that our global society has ever faced.

Many of these issues do not have to be faced in the context of ancillary benefits. Most of the key ancillary costs and benefits are relatively *short-term* and most of the benefits are '*local*' -i.e. they affect the communities relatively close to the source of the policy or program. In both these respects ancillary benefits and costs are very different from the benefits of climate policy, which occur over decades and which have an effect at a global level. However, they are similar and easily compared to the cost of climate policy, which occur locally to jurisdictions making policy.

The advantage of being able to deal with ancillary costs and benefits as local rather than global should not be underestimated. When this is the case, the dramatic cross-country differences in income and ability to pay that plague benefit-cost analysis when applied in an international setting become largely irrelevant. What matters then is the local opportunity cost of climate mitigation compared, in part, to local preferences for better health and other environmental improvements. These opportunity costs will be larger in wealthier nations than poorer ones, but so will the ancillary benefits and costs. In either type of country, they can guide efficient resource allocation and act as a useful input into climate policy decisions.

Furthermore, troubling questions of how to deal with intergeneration and international equity are omnipresent in climate discussions, but they are largely irrelevant for the calculation of ancillary benefits. The ancillary benefits stemming from reductions in conventional pollutants largely accrue in the same timeframe as when cost of climate policies are incurred. (Costs also may be incurred in the future due to different rates of saving and economic growth that result from climate policy). Handily, ancillary benefits can often be offset in a meaningful calculus with costs when considering climate policy without delving into questions of discounting benefits and costs that accrue in the future.

We acknowledge that some ancillary benefits have lasting or long-term implications, such as changes in acidification of ecological systems. In these cases, intergenerational issues remain relevant. However, as many studies have indicated, the lion's share of quantified benefits from reductions in conventional pollutants can be expected in the air-health pathways. Other areas that also have been found important in previous studies, such as transportation-related externalities, also accrue in the present.

We also acknowledge that atmospheric transport of pollution is important, and transport crosses over boundaries separating economies and societies that have different wealth and culture. Usually, however, the dispersion of conventional pollutants is regional or local in nature, and usually at the regional level the similarities outweigh the differences among affected countries, especially when compared to global diversity. Counter examples may be found in Eastern Europe, Southeast Asia and elsewhere, but these do not constitute the major considerations to be faced. Further, global analysis of ancillary benefits has a value in illustrating the potential magnitude of these effects and to ignite interest by researchers and international agencies in the issue.<sup>16</sup> Indeed, we rely on analysis at the global level to illustrate the possibility of leakage of carbon emissions. However, global modeling and

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<sup>16</sup> Davis, *et al.* (1997).

analysis is necessarily weak in describing the institutional setting at the national or sub-national level that plays prominently in the realization of ancillary benefits in practice.

We remain agnostic about the potential ways in which information about ancillary benefits may be used, but the intent for how these estimates will be used will affect the value of additional information, the level of detail and the allocation of resources over aspects of the study. One can imagine ordering possible policy responses to a finding of large ancillary benefits from least to most interventionist. The least, and the one favored by Pearce (2000), is to simply use the existence of large ancillary benefits as an additional rhetorical “no regrets” argument in favor of action on climate change — particularly since such benefits will be experienced in the same time frame as costs. Somewhat more interventionist is to argue for a speedup of climate policy implementation or a spread to additional countries. In developing countries, the consideration of ancillary benefits may affect both the willingness to enter the international control regime and the level of participation.

More interventionist still would be to make the carbon reduction target or the tax more stringent, in light of the greater marginal benefits to be derived for the same marginal cost. Finally, the most interventionist approach would be the idea that the character, perhaps in addition to the stringency, of climate mitigation policy would be affected. If ancillary benefits are significant, it could make sense — say under a carbon trading regime — to concentrate carbon reductions in areas or sectors affecting the most people, holding the carbon reduction target constant. In this way ancillary benefits could be maximized for meeting the given carbon reduction target.

There are a variety of ways that this could be implemented, and a variety of challenges in targeting carbon reduction in this way. The approach depends on the type of carbon policies that are implemented. Under a carbon permit trading system, one way to do this would be to alter the initial allocation of carbon permits with an eye towards maximizing ancillary benefits. A carbon tax system could be designed to spatially or sectorally differentiate taxes to get the most ancillary benefits bang for the carbon reduction buck, again, with an eye to meeting the same carbon reduction (or incurring the same aggregate cost). Under the patchwork of policies that many nations are likely to implement, the consideration of ancillary benefits could play as one factor in the determination of relative burden that will be imposed upon various sectors or regions.

To summarize, we feel the heart of the analysis of ancillary benefits involves the *here and now* that is relevant to individual policy makers in a national context. In this paper we concentrate on the issues and methods we think most important in identifying and measuring ancillary benefits and costs in order to inform national-level policy analysis regarding GHG mitigation. These issues include: (i) consistent definition of costs and benefits, and an illustration of the variety of possible ancillary benefits; (ii) identification of the pollution, policy and population baselines for GHG policies; (iii) description of methods and identification of research breakthroughs that could alter conventional wisdom on the size of these benefits; (iv) the identification of possible ancillary costs of climate change mitigation; (v) identification of the proper scale and scope, and treatment of uncertainty in modeling of economic behavior and physical processes; and, (vi) special considerations in a developing country setting.

## 2. Types of externalities

### 2.1 Definition

Externalities may arise when economic activity has effects on third parties. However, all impacts on third parties may not necessarily be externalities. Such effects may be viewed from a variety of perspectives, including, especially, the perspective of someone who is harmed. However, the term “externalities” applies to an economic analysis focused primarily on an efficiency perspective. We are interested in cases when a policy yields a change in the *productive* use of resources, or in the *welfare* of individuals, and when these effects are not fully taken into account by the agents involved. The magnitude of an externality can be measured by comparing the difference between the *social opportunity cost* of resources that are used in production, and the *private market cost* of those resources.

The focus on ancillary effects means we are focused on effects other than the reduction in GHG emissions, but which occur indirectly as a consequence of those reductions. The Appendix to this paper provides a thorough discussion of the economic concepts involved in identifying externalities, and more complete definitions of some of the terms used in this paper. A couple of the main points to appreciate are the following.

First, externalities do not necessarily arise when there are effects on third parties. In some cases, these effects may already be recognized in, or “internal to,” the price of goods and services. Consider a stylized example, such as damages to vehicles in an automobile accident. If each driver is fully liable for damages to other vehicles and one can reliably assess fault and enforce liability, then the damage in an accident would not be an externality because the party at fault would fully recognize the costs. In this case behavior should reflect the possibility of an accident, *ex ante*, and behavior should be “efficient,” even though accidents would occur with a probability greater than zero. Moreover, one can see a variety of justifications for considering externalities in this example. If drivers are not fully liable, or if fault could not be established, or if liability was not enforceable, then the behavior of one driver could not be expected to fully reflect potential damage to others. Any of these exceptions provide a justification for treating the damage to vehicles in the example as an externality. The key idea to note is that such exceptions constitute a deviation from ideal institutions. In economic vocabulary, this is referred to as *market failure*. For damage to be considered an externality from the viewpoint of economic efficiency, one should be able to identify some kind of failure in markets or other institutions that causes individuals to fail to take into account the social costs and benefits of their individual actions. From a practical perspective, it is also important that such failures result in an important misallocation of resources.

A second point to appreciate is that the economic accounting of externalities does not hinge on the provision of compensation. In fact, there is a reason to deny the provision of compensation (though this remains a controversial point in benefit-cost analysis), because the potentially harmed party may have inadequate incentive to “take care” if compensation is guaranteed. Hence, efficiency analysis leads to findings that may seem irrelevant from an equity standpoint, and *vice versa*. Nonetheless, equity considerations remain important in their own right. Successful public policy must weave together both efficiency and equity considerations.

### 2.2 A partial taxonomy

There are a variety of effects that may result from GHG policies that are secondary to the reduction in GHG emissions. For example, existing studies have identified mortality and morbidity benefits



associated with collateral reductions in particulates, nitrogen dioxides and sulfur dioxides from power plants and mobile sources as a major source of ancillary benefits. Additional areas that might be considered include improvements in ecosystem health (for instance, from reduction in nitrate deposition to estuaries), visibility improvements, reduced materials damages, and reduced crop damages. Reduced private auto use and substitution of mass transit will reduce air pollution and congestion and may also reduce transportation-related fatalities from accidents, although the size of this effect and the degree to which it would count as an ancillary benefit are unclear.<sup>17</sup>

At the same time, there may be ancillary costs of GHG mitigation, such as an increase in indoor air pollution associated with a switch from electricity to household energy sources (such as wood or lignite) or greater reliance on nuclear power with its attendant externalities. In developing countries pollution may rise if electrification slows as a result of policy-induced increases in electricity prices. A related cost would stem from foregoing the benefits of electrification, which include increased productive efficiency and emergence of new technologies, to increases in literacy (Schurr, 1984).

The following offers an illustrative set of examples and an indication about whether they are potential ancillary benefits (+) or costs (-). Note that under certain conditions, any of these observed impacts would not necessarily count as externalities from the standpoint of economic efficiency, depending on whether market or institutions fail to account for these impacts in the incentives they provide for individual behavior.

- Reduction in particle pollution when fossil fuel use is reduced (+).
- Increased availability of recreational sites when reforestation programs are introduced (+).
- Increases in household air pollution relative to a baseline when electrification rates are reduced (-).
- Increases in technological efficiency when new technologies are adopted and unit costs fall (+).
- Increases in welfare when a shift to carbon taxation and a reduction in reduces unemployment (+).
- Reductions in road-use related mortality when a shift from private to public transport takes place (+).
- Reductions in congestion when a shift from private to public transport takes place (+).
- Increases in employment resulting from GHG projects where there is excess supply of labor (+).
- Decline in employment due to decreased economic activity resulting from costs associated with GHG projects (-).

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<sup>17</sup> It is noteworthy that a major study in the early 1990s that considered externalities throughout various fuel cycles for electricity generation in the US concluded that among the single highest valued endpoints (among many specifically defined endpoints) were fatalities associated with rail transport of coal and damage to roadway surfaces beyond those internalized in road fees (Lee *et al.* 1995).

- Savings in household time in poor rural households when fuel wood use is replaced by renewable energy (+).
- Reductions in electricity use resulting from higher electricity prices that cause less use and thereby reduce educational opportunities for children (-).

A taxonomy of the main externalities from air pollution, that were developed in the social cost of electricity studies and that are likely to be relevant to ancillary benefit estimation is provided in Table 1. Note also that not all the impacts can be quantified in monetary terms, although we believe them to be potentially important. The cases that are only partially quantified are indicated by an 'AP'. In addition there are other impacts that have not been quantified at all. We simply do not know whether they are important or not. They have been indicated by an 'NA'. Examples of 'AP' include many impacts on eco-systems, and material damages to cultural buildings. Examples of NA impacts include ozone on forests and heavy metals on eco-systems.

**Table 1.** A sample of externalities assessed in studies of electricity generation

	Health		Materials	Crops	Forests		Amenity 2/	Eco- Systems
	Mortality	Morbidity			Timber	Other		
PM <sub>10</sub>	AM	AM	AP	n.e.	n.e.	n.e.	AM	n.e.
SO <sub>2</sub> 1/	AM	AM	AP	AM	AM	AP	AM	AP
NO <sub>x</sub> 1/	AM	AM	AP	AM	AM	NA	n.e.	AP
Ozone	AM	AM	AP	AM	NA	NA	n.e.	n.e.
Mercury and other heavy metals	AP	AP	n.e.	n.e.	n.e.	n.e.	n.e.	NA
Routine 3/ Operations	AM	AM	n.e.	n.e.	n.e.	n.e.	n.e.	n.e.
Water pollutants4/	n.e.	n.e.	n.e.	n.e.	n.e.	n.e.	AP	AP
Noise	n.e.	NA	n.e.	n.e.	n.e.	n.e.	AM	n.e.

Source: Developed from Markandya and Pavan, 1999.

Notes:

AM: Assessed in monetary terms, at least in some studies.

AP: Assessed in physical terms and possibly partly in monetary terms.

NA: Not assessed, although we believe they may be important.

ne No effect of significance is anticipated.

1. SO<sub>2</sub> and NO<sub>x</sub> include acid deposition impacts.

2. Effects of PM<sub>10</sub>, NO<sub>x</sub> and SO<sub>2</sub> on amenity arise with respect to visibility. In previous studies these have not been found to be significant in Europe, although they are important in the US.

3. Routine operations generate externalities through mining accidents, transport accidents, power generation accidents, construction and dismantling accidents and occupation health impacts. All these involve mortality and morbidity effects.

4. Water pollution effects include impacts of mining (including solid wastes) on ground and surface water, power plant emissions to water bodies, acid deposition and its impacts on lakes and rivers (partly quantified).

### 2.3 *Issues in taxonomy of ancillary effects and the mapping to externalities*

Some of the effects listed above are not externalities in the pure sense, or in every instance when they are observed. An example is employment effects. Employment decisions might be regarded as properly the domain of private decision-making. However, consideration of changes in employment, from an efficiency perspective, is justified in cases when there exist (a) poor information and (b) limited possibilities for these benefits to be represented through market prices.

A comprehensive taxonomy of environmental, public health, agriculture, land use and economic impacts that may qualify as externalities for the purpose of measuring ancillary benefits has been developed in at least three recent studies (Lee et al, 1995; Hagler-Bailly 1995; European Commission 1999). These studies each looked at full fuel cycles associated with electricity generation. The range of effects legitimately range from health and safety issues, to ecological issues and even to economic issues. Many effects map to externalities in a fairly straightforward way; this is certainly true for public health impacts of changes in conventional air pollutants, which are likely to be the single most important category according to these previous studies. However, each impact requires an assessment of whether it qualifies as an externality according to the definitions and criteria we have laid out in this section and the appendix. We illustrate this further with four examples of cases when effects may not qualify as externalities.

- iii. *Occupational health and safety:* Changes in the use of fuels will have implications for the number and severity of workplace injuries. However, there are pre-existing mechanisms through which such injuries may be at least partially internalized in product prices. One way may be through the liability of employers. If workers have the right to sue for compensation for injury then employers must bear some portion of the cost of injuries, and this provides some incentive on their part to reduce workplace risk. Also, wages are likely to vary among employment categories reflecting in part the variation in workplace risk and working conditions. This also serves to internalize some or all of the effect on workers. Consequently, if changes in fuel use lead to a reduction in workplace injury associated with one fuel cycle, this reduction may not necessarily count as an economic externality. The degree to which it should do so is likely to vary across countries and industries.
- iv. *Employment changes:* Changes in fuel use have the potential to reduce jobs in a sector or region of the country resulting in temporary unemployment and transition costs. These changes may not constitute externalities, however. In a fully employed economy, resources will rapidly be transferred to new use. For employment changes to be viewed as externalities, Lee *et al.* 1995 show that these changes must occur in a region that has persistent unemployment due to some type of failure in labor markets. Even in a fully employed economy, the loss of “social capital” and other types of transitional costs can incur resource costs. The degree to which employment changes should be viewed as externalities hinges on a detailed assessment of the labor markets that are affected by a policy.
- v. *Energy security:* Guaranteeing a reliable source of energy has been a central objective of national governments throughout the last century. However, most of the justification for concerns about energy security has stemmed from events outside the normal operation of markets, including cartel behavior and war. Even in the case of supply disruptions, it is well established that a major portion of the resulting economic cost was due to the response of affected countries to supply disruptions, rather than to the actual disruption of energy supply (Bohi, 1984). Furthermore, though the cost of military interventions to maintain oil supply lines has been substantial, it would not be

appropriate to apportion this cost to marginal changes in fuel use because they were largely a fixed cost, with precedent relevant to a variety of contexts. How to analyze energy security is largely a strategic issue, but not one that lends itself well to economic analysis, in our view.

- vi. *Induced technological change.* Induced technological change may or may not be an example of an ancillary benefit. The important principle is consistency in order to avoid double-counting. If technological change is a benefit, the opportunity cost of redirecting resources in new technological directions has a cost that can be significant (Goulder and Schneider, 1996). This cost stems from the loss of value from research and development that was foregone that would have occurred in the absence of the GHG policy and would have yielded a different set of social benefits.

## 2.4 *Issues in framing the analysis of ancillary benefits*

This section introduces two additional issues that would characterize a thorough analysis of ancillary benefits and presents an overall framework for their estimation in graphical form.

### 2.4.1 *Ancillary benefits legitimately vary under alternative climate change mitigation policies*

Estimates of ancillary benefits may vary significantly in different studies or under different GHG policies due to the type of climate change mitigation policy being considered. This can make it difficult to discuss ancillary benefit estimates without qualifying the estimates according to the GHG policy context. Furthermore, when estimates vary, it allows for confusion and provides an opportunity for critics to question the reliability of the methods. However, such variance in estimates may be **legitimate**, and indeed may be a necessary criterion as a measure of the rigor of the methods that are used.

For instance, imagine a policy in the US that targeted the use of coal in electricity generation. This information alone does not provide a meaningful context for evaluation of ancillary benefits, because the policy could have a variety of impacts with respect to emissions of conventional pollutants. If the policy imposed a moratorium on coal development, something that would be sure to have its supporters, it would have a very different effect than a policy that sought to shut down older and less efficient coal-fired power plants. Rowe, Smolinsky, and Lang (1996) point out that emission rates at coal plants in New York State can vary by up to an order of magnitude, depending on their vintage. Heat rates at these plants, which would determine their carbon emissions, are likely to vary by a factor of 2:3. Hence, the change in conventional pollutants per ton of carbon reduced may vary by a factor of six between these two types of policies if they were applied in New York State. If one considers the geographic location of emissions from a national perspective, the likely location of a displaced new plant would affect a rural population with density just a fraction of that in New England. Hence, the ancillary benefits of these two policies may vary by an order of magnitude.

### 2.4.2 *Tax and regulatory interaction effects*

When markets are distorted away from economic efficiency, due to pre-existing taxes or other regulations, the cost of new regulatory policy is affected. The cost of policy (and perhaps the benefit) is likely to be considerably greater than would be indicated by an analysis that does not recognize

these pre-existing features of the economy.<sup>18</sup> Most economists would call this a **direct** (though often unmeasured) economic cost of regulatory policy, and not an ancillary cost.

The relevance of interaction effects to the measure of **ancillary benefits or costs** is two-fold. First, if these costs are excluded from benefit-cost analysis of GHG policies, then they should be accounted for in ancillary benefit studies.<sup>19</sup> Second, these economic costs are not only a characteristic of the GHG policies, but they also characterize other environmental policies in the regulatory baseline. Climate change policy may affect the cost of existing regulation. If the cost of attaining non-climate regulatory goals falls, with it will fall the cost of the tax interaction effect, thereby magnifying the cost savings.<sup>20</sup>

The magnitude of the interaction effect hinges importantly on the type of policy instrument that is used for achieving climate goals. For instance, a carbon tax generates revenues that can be used to offset pre-existing taxes, thereby lessening distortions associated with a new tax or regulation, but typically not erasing them. Estimates of the cost of carbon taxes that include the interaction effect are typically 30 per cent greater than would be estimated if the interaction effect was ignored. A system of tradable permits with permits allocated without charge (grandfathered) can be dramatically more expensive. This is because grandfathered permits impose a cost through interaction with preexisting taxes just as would an environmental tax, but they does not raise revenues that can be used to lessen pre-existing taxes (Goulder, Parry and Williams, 1998). Other authors obtain a lower economic cost if the revenues recovered from a carbon tax are directly specifically toward reform of the most distorting taxes (Jorgenson *et al.* 1995).

The magnitude of interaction effects also depends importantly on the national setting. For instance, Bye and Nyborg (1999) find, for Norway, that the recycling of revenue from a carbon tax can nearly outweigh the cost associated with the tax-interaction effect. The existing literature applies to the US and nations in Europe. Differing rates of unemployment, taxation, and various regulations affecting factor markets affect the results. The measurement of interaction effects in less developed nations is likely to be much more difficult and is a forefront topic in economic research.

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<sup>18</sup> Parry (1995), Goulder (1995). Williams (1999) points out benefits may be greater too.

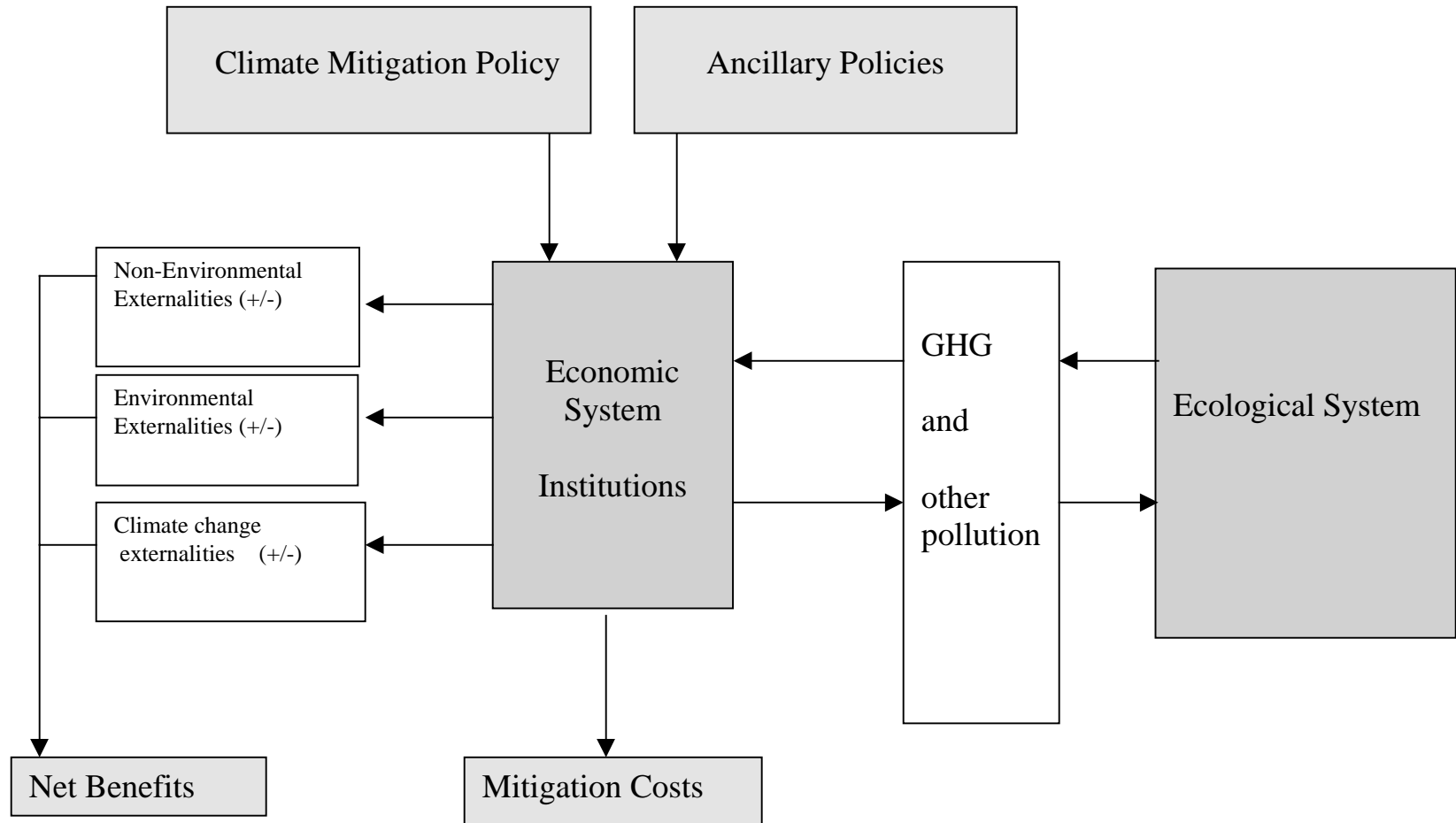
<sup>19</sup> An important measure of tax losses is through the marginal cost of public funds literature. If a mitigation program is funded through increased taxation, or reduces the present burden of taxation, it has a welfare effect measured by the marginal cost of public funds times the changes in taxes. The theoretical discussion of the marginal cost of public funds is surveyed by Hakonsen (1997). Empirical estimates of the marginal costs have been made by the World Bank and others (World Bank, 1997; European Commission, 1998). The EC uses a value of 1.28 for the shadow price of public funds. A similar value will be valid for other OECD countries, but it is likely, however, that a higher value will prevail for developing countries.

<sup>20</sup> For instance, Goulder *et al.* (1999) find that pre-existing taxes raise the social costs of an emissions tax, a performance standard and a technology mandate for reducing NO<sub>x</sub> in the US by a factor of about 27 per cent over the cost of these policies in a model absent pre-existing taxes.

### 2.4.3 *Graphical framework*

Figure 1 provides a graphical representation of the main ideas noted above. Climate mitigation policy operates through an economic and institutional system within a country that leads to reductions in GHGs, changes in other pollutants, and mitigation costs. The emission changes work through an ecological or environmental system that eventually feeds back into the economic system. Then, depending on conditions of the economic system and its institutions, such as labor markets, tax systems, existing environmental and other types of regulations (represented by the box labeled “Ancillary Policies”) these feedbacks may become environmental externalities (such as changes in conventional air or water pollution), non-environmental externalities (such as employment effects) and, of course, climate change externalities (such as leakage of carbon emissions). Ultimately, and from a country’s efficiency perspective only, the net ancillary benefits/costs may be compared to mitigation costs. Note that there are a variety of additional interrelationships that we are omitting from this graphic. An example is that estimated health benefits might be lower if we recognize that a climate change mitigation policy could hold down temperature increases and, therefore, create less ozone than in a model not allowing for this feedback.

Figure 1. Ancillary benefits and costs of climate change mitigation: A conceptual framework



### **3. Baselines**

#### **3.1 *Definition***

One of the most sensitive elements in the analysis of ancillary benefits from a policy is the “counterfactual” -the assumption of what would happen in the absence of the policy, usually termed the “baseline.” The feature that most singularly distinguishes the quality of previous studies of ancillary benefits is the clarity and careful articulation of the baseline, or lack thereof. Since in principle the baseline is a complete picture of one alternative future, it should be specified carefully with respect to its most important characteristics, including those we touch on here.

#### **3.2 *Consistency of baseline treatment***

In considering ancillary benefit estimates in the context of how policy can use this information, a theme we return to repeatedly, it is extremely important to achieve consistency in baseline assumptions among studies that influence the policy debate, or at least to make the implications of their differences explicit.

For example, imagine an analysis of the costs of GHG policy that examines costs in Europe based on an economic model from 1990 but failed to account for changes in energy use that are expected to result from the Second Sulfur Protocol. The Protocol is likely to raise the cost of coal-fired power production and affect the relative cost of coal and other fuels. But the GHG cost analysis would assume a lower cost of coal-fired power in the baseline, and hence it would overestimate the cost of carbon reductions relative to what would obtain if the Second Sulfur Protocol were accounted for. How should an ancillary benefit analysis be framed in this context?

If the ancillary benefit analysis is enlightened with respect to the role of the Second Sulfur Protocol, it would assume lower sulfur emissions in the baseline than are implicit in the GHG cost analysis. Hence, it would conclude the ancillary benefits from changes in sulfur emissions would be relatively low. However, this would impart an important inconsistency in the two analyses and it may provide spurious information for policy-makers who presumably wish to consider total direct and ancillary benefits and costs. Taken together, the two studies would overstate the total cost of GHG policy relative to its benefits. Given the fact the GHG cost analysis erred in its assessment of the regulatory policy baseline, the consistent assumption for the ancillary analysis is the same regulatory baseline. Otherwise, the social opportunity costs and benefits of the GHG policy will be misrepresented.

The importance of the baseline issue is evident in the review of previous studies for the US in Burtraw *et al.* (1999). Though the studies that are reviewed varied for a number of methodological reasons, the most apparent was the treatment (inclusion or not) of the 1990 US Clean Air Act Amendments and especially the tradable permit program for SO<sub>2</sub> (see next section). This difference explained most of the difference in monetary assessment of benefits in the recent studies of ancillary benefits for the US.

Generally, it is unclear which baseline is preferable for the ancillary benefit analysis. It may be that the opportunity costs of GHG policies is affected relatively more by regulatory changes than are the ancillary benefits, or the opposite may be the case. The preferable approach would be to replicate the baseline that is the foundation for studies to be included as a basis for policy, while perhaps also attempting to forecast regulatory changes that might seem important for ancillary benefits.



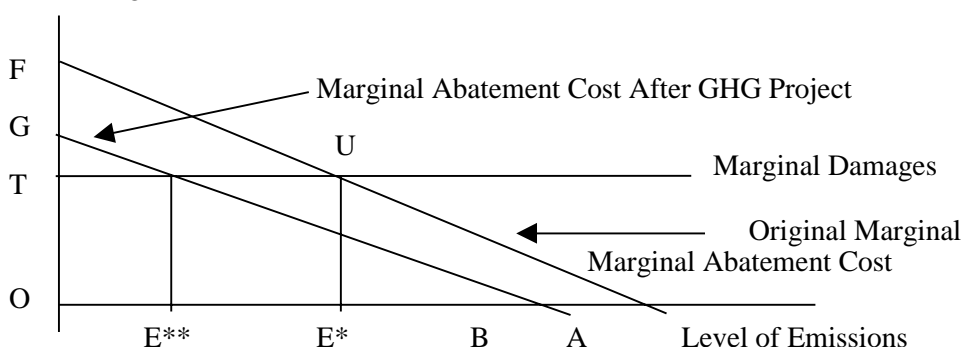
### 3.3 Types of baselines

#### 3.3.1 Regulation of conventional pollutants

In Section III we noted that the relevance of external costs depended on the regulatory framework in place.<sup>21</sup> If the government has taken measures to internalize observable external effects such as damage from conventional air pollutants, then the ancillary effects of GHG reduction may not yield corresponding economic benefit equal to the change in the external cost. The reason is that pre-existing regulation has already incorporated into product prices some portion of external costs. In fact, the policy may create a divergence between the market price and the social cost. The point can be illustrated in Figure 2 below, where we plot marginal damages from fossil fuel emissions and the marginal costs of abatement of those emissions. We assume that the marginal damages are constant and do not vary with the level of emissions (a reasonable assumption in the light of the empirical evidence on this - see e.g. European Commission, 1999).

Figure 2. Ancillary benefit estimation with different regulations

Marginal Damages and Marginal Abatement



Suppose that the government has a tax on the emissions of  $OT$  and the resulting emissions are set at  $OE^*$ . In that case the externality is fully internalized, the government receives a revenue  $OT \cdot OE^*$  and no further adjustment is required. If a GHG reduction policy is now introduced, it will reduce the marginal abatement cost for the fossil fuel from  $AF$  to  $BG$ . What are social benefits of this change? The components are as follows:

- Loss of government revenue.
- Savings in abatement costs.
- Changes in external costs.

In this case the change in the external costs is exactly balanced by the loss of government revenue (both are  $OT \cdot (OE^* - OE^{**})$ ) so the only benefit is the savings in abatement cost. More generally, the tax may not be equal to the marginal damages. In that case it can be shown that the ancillary benefits are given by the change in abatement costs plus the difference between the marginal damages and the tax rate, multiplied by the reduction in emissions (see below).

<sup>21</sup> For applications and background see Morgenstern, 2000; Burtraw *et al.*, 1999; Lutter and Shogren 1999; Burtraw *et al.* 1996; Ekins 1996; Freeman *et al.* 1992.

Now consider the case where there are tradable permits for the conventional pollutant and the limit is set at  $OE^*$ . If this limit is unchanged and if the GHG reduction is not so severe that emissions of this pollutant fall below  $OE^*$ , then there is no ancillary environmental effect associated with the GHG reduction. Although there is no ancillary benefit from that source, there is likely to be a decline in the cost of abating the conventional pollutant and this yields economic savings to consumers.

Finally consider the most common case where the regulation is not in the form of a charge but a command and control regulation that enforces a particular emission rate or technology standard. Imagine the regulation sets the emission rate such that emissions are  $E^*$  and the emission rate standard or technology standard is maintained them at that level. As a consequence of subsequent GHG reductions, society benefits to the value of the reduction in conventional pollutant emissions multiplied by the marginal damage, plus the savings in abatement costs.

What the above analysis illustrates is that the estimation of ancillary benefits has to take account of the regulatory framework. The above examples are not exhaustive but provide a range of possibilities. To summarize, the ancillary benefits are estimated as shown in Table 2.

Table 2. **Ancillary benefits under different regulations**

Regulatory Framework(*)	Ancillary Benefits
Emissions charges	$\Delta AC + (MD - T) \Delta E$
Command & Control	$\Delta AC + MD \Delta E$
Permits	$\Delta AC$

(\*) Assumes that regulation of conventional pollutant does not change as a result of the climate policy.

$\Delta AC$ : Change in abatement costs related to the pollutant as a result of the project.

$MD$ : Marginal damages from the pollutant (\$/ton).

$T$ : Charge rate for emissions (\$/ton).

$\Delta E$ : Change in emissions.

### 3.3.2 *Economic and energy regulation*

Energy industries are regulated in a plethora of ways, sometimes even within one country. The differences can have a large bearing on the ancillary effects of GHG policy. For example, state owned electricity companies or companies operating under cost of service regulation may not have an incentive to minimize costs. Deregulated companies, on the other hand, may have such an incentive, but they may also have the opportunity to behave strategically by withholding supply to affect market prices. All these settings have different implications for the change in conventional pollutants that would result from GHG policy. For instance, state owned enterprise may provide subsidies to certain sectors or governments may pass responsibility for GHG reduction on to privately owned industry. Deregulated or unregulated companies may have lower emissions in the baseline (Oates and Strassmann, 1984). Moreover, energy subsidies may take a variety of forms that can affect the response to GHG policy. A most important feature in policy analysis is to identify these baselines, and to identify trends such as the changes in regulation. Finally, repeating the theme mentioned previously, it is important to maintain consistency in the forecasts of changes in the regulatory baseline with other studies of the costs of GHG policy.

### 3.3.3 *Socioeconomic and demographic baseline*

Changes in socioeconomic and demographic information over time can have at least two important effects. Increases in population and income can lead to changes in demand for energy and in emissions

that cause marginal abatement costs for conventional pollutants to increase. Offsetting this to some degree is the rate of technological change (Lutter, 1999). On the benefit side, changes in population and income can lead to greater willingness to pay for environmental amenities, and larger benefits from reduction in conventional pollution (Krutilla, 1968). Also, improved income and other factors can improve health, which can lower baseline mortality rates. Lower rates will lower the benefits from conventional air pollutant reductions. Therefore, it is important to forecast changes in the socioeconomic and demographic baselines in the locales where ancillary benefits are to be estimated. The analysis of transportation related ancillary benefits and costs would seem to be particularly sensitive to the socioeconomic and demographic baseline.

### 3.3.4 *Environmental baseline*

Some environmental issues exhibit thresholds or other non-linearities that imply that benefits do not move directly with reductions in ancillary pollutants. Acidification is an interesting example because damage may result only after critical load thresholds are violated. On the other hand, recovery may not occur with a reduction in conventional pollutants, until some new threshold is achieved or after a significant passage of time.

### 3.4 *Lingering issues in setting the baseline*

This review has emphasized the importance of baseline issues but it is not always possible to suggest resolution to these issues, and this should be a matter of ongoing research. One lingering concern stems from the fact that the IPCC baselines lack appropriate detail for consistent modeling of population and there is no guidance with respect to regulatory baselines. Another concern is that the set of existing and potential regulations is vast, and it is not necessarily clear *a priori* which are the most important to model carefully. In addition, assessment of the environmental baseline is very difficult (e.g., nitrogen saturation, traffic congestion). Investigators should attempt to conduct sensitivity analysis to explore the importance of regulatory baselines and others that may seem most important to the conclusions of specific studies.

Finally, we note that a fundamental characteristic of the way we propose to measure ancillary benefits treats existing regulation, or some portrait of expected future emissions, as a fixed baseline. This baseline may not reflect optimal levels of control, as we have discussed previously. In addition, this baseline may not reflect optimal control were climate and conventional pollutants controlled jointly. The decision to reduce GHG will change the opportunity cost of reducing conventional pollutants, and vice versa. Compared to our formulation of the baseline, joint optimization would lead to lower costs, and hence this approach may be preferred by some analysts. This approach also introduces the cost and/or benefit allocation problem (Austin, *et al.*, 1997). The problem is to determine what portion of cost or benefit is to be allocated to reduction of GHG and what portion to conventional pollutants. We recognize that these policies are inextricably linked over the long run. However, our focus on the near-term in considering ancillary benefits provides a justification for identifying a fixed baseline that is relevant for a timeframe over which those emissions would not be likely to change, absent a change in climate policy.

## 4. **Estimating Ancillary Benefits**

New information is more valuable when it is likely that it will change decisions, in this case decisions about climate policy. If information could be generated that would be very likely to make existing ancillary benefit estimates much larger or much smaller than we find in current studies, such

information would be valuable. Thus, in this section, we employ a value of information approach to highlight, based on admittedly subjective judgements, key issues in the estimation of the benefits of environmental improvements — the raw material of ancillary benefits analyses — that, were they informed by new research results, could change the nature of the ancillary benefits debate. We use our informed speculation to address potential research findings with respect to physical effects and valuation for each of the major categories of benefits. To make the discussion manageable, we confine ourselves to a developed country context and to an air pollution context. For this and other reasons, this discussion is meant to be suggestive rather than exhaustive. We hold aside those of the issues discussed above that are specific to the ancillary benefit debate, such as the effects of the regulatory baseline and the choice of climate policy on such ancillary benefits.

We start from the observation that ancillary benefit estimates range widely as a fraction of climate change mitigation costs. Burtraw et al (1999) show that the highest estimates can be reconciled to some extent with lower ones once baseline issues are addressed. But, the reconciled estimates are still a large fraction of abatement costs and there is significant skepticism about whether such values are realistic. Thus, we take it as axiomatic that new studies confirming findings from the existing literature would serve to further legitimize the view that ancillary benefits are “large.” Beyond this, we want to examine what new research might find that would make such benefits even larger or much smaller.

#### **4.1 Mortality Benefits**

Mortality benefits drive ancillary benefit estimates. They are probably the most studied endpoint and clearly contribute the largest share of benefits. Some studies of the benefits of environmental improvements have found them to be several percentage points of GNP, for instance. There is, however, considerable controversy about these values and it is possible that new information could lead to far lower estimates. The following are some of the key issues that are being discussed that could lead to lower estimates.

- Most epidemiological studies have assumed, but few have searched for, the absence of thresholds in the concentration-response functions. As the bulk of the air pollution distribution is at low concentrations (in developed countries), finding thresholds could dramatically lower estimates of lives saved.
- Incorporating results from some recent studies that show synergies and interactions between pollutants could result in increased or decreased marginal damages being associated with a particular pollutant, depending on how the interaction effects work. Such effects also draw attention to the need to attribute damage by pollutant rather carefully.
- Findings that only a few days-months of life expectancy were added from pollution reductions or that the bulk of life expectancy changes occurred in the future (e.g., the latency effects from exposure to carcinogens) would result in much lower estimates of life-years saved and benefits.
- Direct particulate emissions are a minor part of the PM<sub>10</sub> inventory. Secondary particulates species-nitrates and sulfates — and road dust make up a large fraction of the anthropogenic PM<sub>10</sub> concentrations. While evidence exists for the role of sulfates in mortality risks, there is very little evidence on the role of nitrates and dust. If these were found not to affect health, estimates of lives saved would fall.

- Perennial debates occur about possible confounders in regression analyses providing the concentration-response functions. Discovering that these confounders are responsible for the observed effects of pollution would lower benefit estimates.
- New information showing that the WTP for reduced mortality risks is much lower for older people and people in ill health would lower benefit estimates

What new information would lead to larger estimates?

- The key mortality studies do not apply to neonatal effects; yet a growing literature finds effects of particulates on mortality of this cohort. Acceptance of these effects as real could dramatically increase benefits.

#### **4.2 *Chronic morbidity benefits***

These are benefits arising from the reduction in long-term respiratory illness and heart disease. What new information would lead to far different estimates?

- Life expectancy might be found to significantly increase indirectly through reduced disease incidence, leading to larger benefits. Double-counting these as mortality benefits might be an issue, however.
- With only a few epidemiological studies on these endpoints and two WTP studies valuing changes in chronic respiratory disease (these surveyed, using a fairly untested approach, only 300 people each), any new information could dramatically alter benefits.

#### **4.3 *Benefits to ecological resources***

The most controversial area for estimating the benefits of environmental improvements is benefits to ecological services, and of these, the values held by individuals that are not tied to use of the resource — i.e., nonuse values. Currently, it is fair to assume that such benefits are listed as unquantifiable, although attempts have been made to value them using a questionnaire, or ‘contingent valuation’ approach.

What new information would lead to such values being taken seriously?

- Ecological studies would become available linking marginal pollution changes to changes in ecological resources.
- The “warm glow” hypothesis or related hypotheses questioning the credibility of nonuse value elicitation would be shown to be wrong.
- Studies would be completed that found WTP for marginal changes in ecological resources and such studies would show sensitivity to scope and meet other tests called for by the NOAA Panel (which laid down the conditions that contingent valuation studies must satisfy if they are to be regarded as credible).

#### **4.4 *Materials damage***

This area is rarely included in benefit analyses. Engineering studies of damage from pollution to buildings, fabrics, monuments, etc. suggest that effects, although notable, are dominated by health effects. However, the studies do not adequately cover cultural and historic monuments and their impacts.

What new information would lead to such values being significant?

- Major database effort to collect information on the inventory of sensitive materials.
- Studies showing that normal maintenance and replacement schedules would be significantly lengthened by reducing pollution.
- Studies showing the WTP for marginally slower degradation of monuments is high.

#### **4.5 *Visibility***

Only in the U.S. has this endpoint been taken seriously, although the many problems with existing WTP studies have recently led to this endpoint being dropped from the peer-reviewed Cost-Benefit Analysis of the Clean Air Act, conducted by the USEPA.

What new information would lead to such values being significant?

- Studies showing that humans perceive the kind of changes to visibility in urban area that could be expected from ancillary SO<sub>2</sub> reductions.
- Studies showing people in these areas are willing to pay significantly for such improved visibility.

#### **4.6 *Crops and tree farming***

Some studies of the benefits of reduced ambient ozone concentrations show sizable increases in yields and social welfare.

What new information would lead to far different estimates?

- Only a few crops and trees have been studied. Studies implicating ozone (or other pollutants) in other crops could raise ancillary benefit estimates.
- The key concentration-yield relationships are engineering-based and do not take into account the joint effects on farming behavior of reduced climate effects from BAU and reduced air pollution. Studies that could do this might show that these effects work synergistically or antagonistically and, in any event, could greatly alter the engineering-based benefit estimates. (There are some general equilibrium studies of crop effects, which show that the prices of crops can alter significantly when emissions levels change. This results in a change in the distribution of benefits between consumers and producers, but not in a major change in total consumer and producer surplus).

## 5. Ancillary Costs (i.e., negative ancillary benefits)

For all the uncertainties associated with linking *given* changes in emissions or other externality-causing actions to health and other categories of externalities, perhaps the greatest uncertainty is the most basic: *Will climate change mitigation policies lead to net increases or decreases in emissions and other externality-causing activities?* Further, will the spatial distribution of what would otherwise be net decreases in emissions result in net ancillary costs rather than net ancillary benefits?

First, the most direct effect of, say, a carbon trading policy involving Annex I countries, would be to reduce use of carbon-intensive inputs (e.g., coal) and production of carbon-intensive outputs (e.g., electricity) through pushing up prices for these types of products. Emissions of conventional pollutants and other pollutants tied to coal and electricity would fall, generating ancillary benefits. However, one must be careful to note that reduction in conventional pollutants has some perverse consequences (Wiener, 1995). Climate models capture the regional cooling that is associated with sulfates in the atmosphere. And, reduction in ground-level ozone may lead to increased exposure to UV-B radiation, especially near the equator (Bruhl and Crutzen, 1989).

Second, reduction in output of carbon-intensive commodities also could be associated with a variety of undesirable effects. Elsewhere we have pointed to the possibility of boosting employment through GHG mitigation policies. But it is also possible that employment could fall. Moreover, though the literature is controversial, a change in employment has been associated with changes in mental health, alcoholism, suicide and spouse abuse. Similarly, a change in income has been associated with other aspects of health status (Viscusi, 1994; Perkins, 1998; Lutter and Morrall, 1994; Portney and Stavins, 1994). Consequently, the negative impacts on employment or income may have social consequences that are not captured in economic models. We acknowledge that positive impacts also may have commensurate positive social consequences. Our point is simply that consideration of employment changes is a two-edged sword. Usually this effect is left out of economic models under the assumption of full employment. So when these models are criticized for failing to capture employment changes, one must recognize that the effect could point in the direction of benefits or costs.

Third, output reduction of coal-fired electricity will lead to substitution toward low carbon substitutes, which are not necessarily low in causing externalities. One example is the substitution to nuclear power in place of coal for electricity generation, which would have attendant health and other types of risks. A switch to hydroelectric power could create many negative externalities to river ecosystems. Another example would be a switch to diesel for transportation fuel, which would have a lower carbon content than gasoline but would have greater emissions of conventional pollutants.

Fourth, reduction in the use of electricity could lead to substitution toward other unmonitored fuels that may increase externalities. A potentially important example of output substitution is in home fuel use (particularly in developing countries), where a reasonable consequence of a *global* trading scenario is for an increase in indoor air pollution associated with a switch from electricity to dirtier household energy sources such as wood or lignite. This may have tremendous significance in specific locales where indoor air pollution is a major health risk, and where delays in electrification also mean delays in attainment of literacy. However, we acknowledge that in most developing nations the institutional failures in delivering and charging for electricity services pose a larger barrier to electrification than would carbon policy. More generally though, pollution or other adverse consequences may rise if electrification slows as a result of policy-induced increases in electricity prices.

Fifth, and related to the above, may be the “sink” effect. If carbon policies encourage the use of sinks for energy use, these sinks may have significant negative externalities relative to the coal they replace. Large tree farms, for instance, may create damages to ecosystems because of their reliance on monoculture. While unambiguously better from a carbon perspective, ancillary benefits may not be positive.

Sixth, there might be an “ancillary leakage effect.” Imagine a carbon tax for Annex I countries that leaves carbon use in other countries uncontrolled. Such a policy would drive a wedge between demand and supply for coal and oil in Annex I countries, with a reduction in world coal demand forcing down the coal price from the perspective of non-Annex I countries. This leads to an increase in the use of coal because of the change in relative factor prices, and it leads to an expanded export market for goods in non-Annex I countries because of their relative cost advantage. (This is offset by potential shrinkage in demand in Annex I countries, whose economies are incurring costs associated with GHG policies.) The resulting increase in coal use (and in use of other fossil fuels) in non-Annex I countries — the carbon leakage — brings with it an ancillary cost of greater air pollution and other negative externalities. Because control efficiencies of conventional pollutants are lower in developing countries than in developed countries, and, perhaps, population densities near power plants and other large users of energy may be larger in developing countries, ancillary costs may be larger than suggested by carbon leakage or fuel use changes (Wiener, 1995).

To put a bit more perspective on this issue, we consulted two published articles modeling the effect of Kyoto and various forms of carbon trading on carbon leakage and energy production and consumption (Bernstein et al, 1999; McKibben et al, 1999) and obtained new runs of an improved Bernstein et al model. (See Table 3.) McKibben et al find, under a scenario where Annex I countries meet their Kyoto commitments through autarky without carbon trading among them, that energy consumption, particularly of coal, falls dramatically in the Annex I countries. In contrast, LDCs (excluding China) increase coal consumption by 0.3%, oil consumption by 5.1%, and gas consumption by 3.4% in 2010. However, consumption of these fuels falls in China by 0.8%, 0.4% and 1.2%, respectively.<sup>22</sup> As for carbon leakage, Annex I base carbon is 3,644 million tonnes (excluding FSU), China is 1589 million tonnes, and LDCs are 2392 million tonnes. Changes under no trading are: -1102, -12, +79 million tonnes, respectively, for 6% carbon leakage overall.

Turning to the Bernstein et al results, the coal reductions in Annex I countries are very similar to those in McKibben, although drops in oil and gas use are different. Coal consumption falls the most in percentage terms, average around 50%. Electricity consumption falls as well — 31% in the U.S., and a far smaller amount (12-19%) in other developed countries. At the same time, because of the drop in demand for these fuels, their prices fall. This leads to a ‘bounceback’ effect on energy consumption in the FSU as well as in the developing world, or what we have termed the ancillary leakage effect. Coal consumption rises by 4% in the FSU and 5% in the developing world. Increases in consumption of other fuels are commensurate. Electricity demand in the FSU actually increases 13%. In the developing world, electricity consumption is flat or rises slightly. This change in the use of coal is far larger than that predicted by McKibben *et al.* China and India’s use of coal increases almost 1% in this study, compared to a reduction of almost 1% in McKibben *et al.*, and oil use increases almost 4% compared to a -0.4% change predicted in the McKibben *et al.* model.

Though the McKibben *et al.* and Bernstein *et al.* results differ in important ways, they share a common finding with respect to the possibility for severe leakage out of the carbon regime. In the relatively short run, while the international regime excludes important carbon sources, this suggests the

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<sup>22</sup> The introduction of trading among Annex I countries does not have a large effect on these percentages in China and the LDCs, but fuel consumption falls by much less in the Annex I countries.



unfortunate possibility of significant ancillary costs that could result from changes in conventional pollutants in certain regions as a consequence of carbon reductions under the Kyoto Protocol.

Table 3. **Illustrations of carbon leakage in modeling exercises**

<b>Percent Change Energy Consumption from Baseline (2010)</b>				
	Coal	Oil	Gas	Electricity*
USA	-48.2	-18.0	-29.0	-30.7
Japan	-59.3	-7.2	-48.8	-12.1
Europe	-43.1	-1.2	-27.4	-11.3
Other OECD	-43.1	-6.4	-40.4	-19.6
Former Soviet Union	3.8	4.0	12.6	12.7
Developing Countries	4.8	3.7	4.8	-

*Source:* Montgomery (2000).

<b>Percent Change Energy Consumption from Baseline (2010)</b>				
	Coal	Oil	Gas	Electricity
Europe	-51.4	-2.9	-7.6	-4.7
North America	-65.5	-15.0	-11.9	-16.8
Japan	-60.8	-14.2	7.0	-11.4
Other OECD	-53.0	-2.9	7.7	-5.1
Former Soviet Union	3.1	4.5	2.2	3.0
Developing Countries	5.6	3.7	-0.5	1.9

*Source:* Detailed Results from Bernstein et al (1999).

<b>Percent Change Energy Consumption from Baseline (2010)</b>			
	Coal	Oil	Gas
USA	-51.9	-15.6	-12.6
Japan	-43.6	-14.2	-4.6
Australia	-55.1	-18.4	-19.4
Other OECD	-49.6	-29.5	-18.2
China	-0.8	-0.4	-1.2
LDC (All countries other than OECD and China)	0.3	5.1	3.4

*Source:* McKibbin et al 1999.

Finally, Lutter and Shogren (1999) point out that ancillary costs could arise from the geographical reallocation of economic activity following a carbon mitigation policy. If carbon trading were in place, for instance, some areas, relative to their carbon allocation baseline, would be net sellers, others net buyers. In extreme cases, some net buyers could actually exceed their BAU carbon and conventional pollutant levels. Such cases may be far fetched. However, less far fetched is the possibility that net carbon permit buyers are near urban areas, while net sellers are not. In this case, population exposures to ancillary pollutants could increase on net, even with constant aggregate carbon emissions.

It is interesting to note that the examples of ancillary costs given above relate to ‘macroeconomic’ policy options rather than ‘micro’ decisions, where investments to replace carbon generating technologies are being considered. Although ancillary costs could also arise in such cases, they are less likely to be as significant. This underscores the point that the kinds of ancillary costs and benefits considered depend on the policies being evaluated and a specific national and institutional context.

## 6. Other general issues

### 6.1 *Issues of scale/space*

We have stated above that ancillary benefit estimation is primarily a country-level matter from a policy perspective, since individual countries will decide on how to achieve their agreed commitment to carbon reductions based on an assessment of the costs and the ancillary benefits of alternative actions. A related proposition is the fact that *estimates should vary in different applications*. Indeed, a criterion for evaluating the credibility of previous studies should be the way in which they vary, depending on issues of scale and space.

The series of studies on the social cost of electricity (Lee *et al*, 1995; European Commission, 1999; Hagler Bailly Consulting, Inc. 1995), and studies such as Burtraw et al (1999), make it clear that credible estimation of benefits from reductions in pollution requires modeling at the local level. The size of benefits depends on where the emissions are in relation to “receptors,” the people, economic resources, and ecological resources susceptible to pollution exposures. And the extent and exact spatial distribution of effects is determined by physical features of the medium distributing the pollution, i.e., air, stream, groundwater flows, as well as other features of the “landscape,” such as mountains, temperature, rainfall frequency, and the like.

The importance of space in determining benefits implies that economic activity will also have to be modeled at a disaggregate, spatial level. The Burtraw et al paper is probably at the limit of the aggregation over space that preserves reasonable spatial detail for benefit estimation. In this paper, U.S. electricity supply is modeled at a multi-state level (thirteen NERC regions), and then apportioned out to specific plant locations according to historic generation rates. The emission data is then aggregated back up to the state level and married to a set of source-receptor coefficients for SO<sub>2</sub> and NO<sub>x</sub> emissions converted to PM<sub>10</sub> concentrations specified at a state level to estimate health effects at that level. As seen in Krupnick and Burtraw (1996), which contrasts the output of several of the social cost studies, fine details of plant specification — stack heights, exit gas temperatures, type of fuel burned (high or low sulfur coal, for instance) abatement technology in place in the baseline — can make a big difference in the estimate of benefits, along with the features of the receptors (e.g. population demography, visitation to recreational sites, use of various impacted streams by fisherman; catch rates).

This degree of disaggregation may be contrasted with that typically, and appropriately, used to model sector level responses to various policies to combat global warming. Generally the computable general equilibrium (CGE) models are specified without spatial detail. Indeed, it is almost surely too much to expect that such models be designed to incorporate this type of detail without compromising their usefulness to shed light on the costs and other economic consequences of climate change mitigation. Hence, the analysis of ancillary benefits to climate change policy invites a disaggregated or local modeling strategy to complement the aggregated large scale or CGE modeling necessary to integrate economic relationships.

### 6.2 *Treatment of uncertainty*

There is general agreement that the uncertainty surrounding the estimates of ancillary benefits and costs is at least as great relative to the value of those estimates as that associated with other mitigation costs. The process by which external costs and benefits are calculated involve a number of physical modeling steps and a valuation step. The modeling involves estimation of emissions, their dispersion and transformation, and the impacts of the pollutants. The valuation of the impacts is based on statistical techniques that also have large error bounds. Each of the steps also has some uncertainty

associated with it in terms of modeling choices. And the cumulative uncertainty, which is a combination of model and statistical uncertainty, could be quite large.

The first point to note is that a good study of ancillary costs and benefits will provide some idea of how large the statistical uncertainty bounds are. A single number is indicative of a misleading approach and of less than thorough analysis. The second point is that there is more than one way to report the uncertainty. For the statistical uncertainties, it is possible to derive probability intervals, using Monte Carlo methods, or by other statistical methods. For model uncertainty other methods such as bounding analysis, breakeven analysis or meta analyses have been used. Finally a method that integrates both types of uncertainty based on subjective and objective error estimates is that of Rabl and Spadaro (1998). This method provides a quantification of the uncertainty and, recognizing that many studies do not have enough information to carry out a quantitative analysis, reports a subjective qualitative indicator of uncertainty. For climate change work, Moss and Schneider (1998) suggest that model uncertainty be described as follows:

- “Well Established”: models incorporate known processes; observations consistent with the models; multiple lines of evidence support the cost assessment.
- “Well posed debate”: different model representations account for different aspects of observation/evidence, or incorporate different aspects of key processes, leading to different answers. Large bodies of evidence support a number of competing explanations.
- “Fair”: models incorporate most known processes, although some parameterisations may not be tested representations; observations are somewhat inconsistent but are incomplete. Current empirical estimates are well founded, but the possibility of changes in governing processes is considerable. Possibly only a few lines of evidence support the evaluations.
- “Speculative”: conceptually plausible ideas that have not received much attention in the literature or that are laced with difficult to reduce uncertainties.

At the least, ancillary benefit studies should provide similar qualitative information.

## **7. Developing country issues<sup>23</sup>**

### **7.1 Definition**

Developing Countries (DC's) cover a wide variety of countries with distinct differences in terms of the economic, political, social and technological levels. The group of countries termed “Least Developing Countries” have very little basic infrastructure, the “Newly Industrialized Countries”, have a structure closer to that of the industrialized countries, and others lie between these two extremes.

Since GHG limitation does not have as high a priority relative to other goals, such as poverty reduction, employment, etc. as it does in the wealthier countries the issues of ancillary benefit estimation are all the more important. Indeed, one can argue that the major focus of policy will be development, poverty alleviation etc. and that GHG limitation will be an *addendum* to a program

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<sup>23</sup> This draws in part on the contributions of writing team for Chapter 7 of the IPCC TAR on Cost Methods.

designed to meet those needs. Taking account of the GHG component may change the detailed design of a policy or program, rather than being the main issue that determines the policy.<sup>24</sup>

Developing countries in general exhibit a number of specific complexities that raise further difficulties or that need even more attention than in developed countries when estimating ancillary costs/benefits. Data are limited, exchange processes are constrained, markets incomplete, and a number of broader social development concerns need to be taken into account, such as living conditions of the poor, gender issues, and institutional capacity needs. Some of these difficulties arise particularly in relation to land use sectors, but can also be important to consider in relation to the energy sector and transportation. Because of these problems, a simplified application of methodologies in developing countries can lead to a number of inaccuracies in ancillary benefits studies. We discuss four key issues that need to be addressed in developing countries studies.

## 7.2 *Availability of adequate local valuation studies of external effects*

The estimates of external effects in developed countries are derived from spatial modeling of pollutants and their impacts and from valuation studies that elicit local WTP/WTA. In many developing countries such studies are not available, although the number and quality of studies is improving (Krupnick, Davis and Thurston, 2000)<sup>25</sup>. Where there is a lack of local information on WTP, one option is to use studies from developed countries and ‘adjust’ the estimates for local conditions. This procedure is called benefit transfer, which is defined as “an application of monetary values from a particular valuation study to an alternative or secondary policy decision setting, often in another geographic area than the one where the original study was performed” (Navrud (1994)). The problems of such transfers are discussed in greater detail elsewhere (see Davis, Krupnick and Thurston, 2000). In some respects, damages associated with activities such as transportation may be greater in developing countries, due to the greater reliance on more polluting varieties (e.g. motor bikes), poor conditions of vehicles etc. This has to be taken into account in making any benefit transfer from studies in developed countries.

Another option is to “retreat to defensible borders.” At least with respect to health effects, probably the major quantifiable ancillary impact, estimates of medical costs for acute and chronic morbidity and of the value of wages lost from premature death (the human capital approach to valuing mortality risk

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<sup>24</sup> For example, Markandya and Boyd (1999) and Halsnæs and Markandya (1999) have examined a number of carbon mitigation projects in developing countries including renewable energy options (biogas, solar water heating systems, PV streetlights, and wind turbines), Demand Side Management Programs, and a number of transportation sector options. An expanded assessment that includes ancillary benefits includes specific valuation of welfare impacts of increased employment, local environmental improvements related to reduced non-GHG pollutants and income distribution weights. From these applications one can conclude that an expanded cost assessment framework has major implications on the cost effectiveness ranking of mitigation projects compared with a focus on direct costs. Big differences in cost effectiveness in particular are seen for a biogas plant in Tanzania, where social costs considered in the expanded framework go down to minus 30\$ per ton CO<sub>2</sub> reduction compared with a financial cost of plus 20\$ per ton CO<sub>2</sub>. This cost difference reflects a positive welfare impact on presently unemployed low-income families and time savings due to reduced fuel wood collection. The case examples generally suggest that social costs of mitigation policies in developing countries in particular will be lower than financial costs in cases, where the policies require presently unemployed labor and are reducing the damages of local non-GHG pollutants.

<sup>25</sup> The lack of studies is a problem not only for the valuation of impacts but also for the dose response functions linking pollutants to physical impacts. Such dose-response studies are also limited. Exceptions are to be found in Cropper et al (1997) (India), Ostro et al (1994, 1997, 1998), (Chile, Indonesia Thailand).

reductions) can be obtained. While these estimates are clearly lower bounds to the value of avoiding or reducing such effects, they will not be as controversial as WTP measures and for many decisions, may be large enough compared to cost to render better information nearly worthless.

More research is needed on estimating WTP in developing countries and comparing such estimates to benefit transfers to better assess the reliability of the latter approach.

### **7.3      *Development projections***

The establishment of long-term projections for ancillary emissions is complicated and uncertain for developing countries. These economies often are in a transition process where such emissions are expected to decrease after a certain level of development, such as in accordance with the environmental Kuznets curve. It is not possible, however, to project accurately the actual speed of this process. Modeling tools and data are also very limited or even non-existent in these countries and the only available information sources for generating such projections are often the official national development plans that only cover a 5 to 10 years time horizon.

The basic uncertainty of long-term GHG as well as ancillary emissions projections encourages one to consider the use of multiple baselines, each corresponding to a particular expectation of the future development pattern. Each development pattern may exhibit a unique emissions trajectory. A nation following development policies which emphasizes greater investments in infrastructure such as efficient rail transport, renewable energy technologies and energy efficiency improvements would exhibit a low emissions trajectory. On the other hand, a nation with substantial coal resources, scarce capital and a low level of trade can get pushed towards a development path with high emissions. In the former case, climate change mitigation would have smaller ancillary benefits than in the latter case.

The spatial distribution of the population and economic activities is very different from that of developed countries, with higher concentrations in urban areas and less suburban sprawl. This high concentration, combined with poor pollution control, result in extremely high pollution levels in such cities and therefore a greater potential benefits from reducing emissions from point and non-point sources close to or inside the cities.

### **7.4      *Employment***

Unemployment or underemployment, especially of unskilled labor is much more of a concern in developing countries than in developed ones. The best way to include employment factors is to apply a shadow price to labor, which reduces the economic cost of employing workers below the financial cost. The actual value for the shadow price depends on the opportunity cost of the worker's time, taxes, and unemployment payments etc. Details of how the shadow price can be calculated for a climate change mitigation project are given in Markandya (1998). Some illustrative examples show that using such shadow prices changes the ranking of labor intensive projects relative to capital intensive ones by a significant margin.

Shadow pricing is normally applied to the direct labor used in a mitigation program or policy. Where there are broader macroeconomic employment benefits, (e.g. through a switch of carbon taxation for taxation on employment), policy-makers generally want to be provided that information along with other indicators of benefits, such as changes in GDP, welfare 'equivalent variation' measures etc. There is little gain in trying to include the employment benefits into a single measure.

## 7.5 *Equity*

A key issue in evaluating climate change policies in developing countries is their impact on individuals differentiated by wealth, and on countries by their level of *per capita* income. For decision-making at the country level — and given that countries are sovereign, all real decision-making is at the country level — values based on local preferences, medical costs, or taken from some other local source would appear appropriate, and they should be unadjusted for income differentials vis-à-vis other countries. Climate mitigation costs are also local and should be compared with these ancillary benefits if the country is to make efficient decisions about climate change policy. A country acting in its own self-interest is free, of course, to consider equity issues in its own decision-making.

Where the *context* demands that ancillary benefits be aggregated to a global or multi-regional level, three approaches have been suggested to deal with the country-level equity issue. One is to report the income changes as supplementary information and allow policy-makers to decide on what weight to place on them. The second is to use ‘income weights’ so that impacts on individuals with low incomes are given greater weight than individuals with high incomes and the third is to use average damage estimates and apply them to all individuals impacted, irrespective of their actual WTP<sup>26</sup>.

Third, it is considered unacceptable by many to impose different values for a policy that has to be international in scope and decided by the international community. In these circumstances, analysts to use the average damage value method. The analyst estimates the money value of impacts for different groups of individuals or countries and then applies the average damage to all individuals and countries. The best example of this is the value attached to changes in the risk of death. On the basis of EU/US values of statistical life and a typical value for the inequality aversion parameter Eyre *et al* (1998) estimate the average world value of statistical life at around one million Euros (approximately one million US dollars at 1999 exchange rates)<sup>27</sup>.

## 8. **Conclusions**

There has been an explosion of interest in the potential for the ancillary benefits of climate change policy to offset some of the costs of reducing greenhouse gases. We define ancillary benefits to be the effects of a climate change mitigation policy other than those related directly to meeting climate change goals, that would not otherwise occur, and that are not internalized in market behavior. If the list of such effects is long and the benefits from each are large, climate policy will look like a far better deal than were these benefits not considered. Indeed, if these benefits are large enough, or concentrated enough in several places, perhaps even the shape and stringency of climate policy should be altered.

The central element of this story is that policies to abate or otherwise reduce GHGs will lead to reduced energy use, which will reduce conventional air pollutants along with it, bringing large improvements in health, visibility, crop yields, and other benefits linked to air pollutants. Additional benefits categories form a long list, including reduced traffic accidents and fatalities from lower vehicle-miles-traveled, reduced soil loss from increased tree farming, even reduced unemployment where labor is in excess supply. For developing countries that have difficulty mounting anti-pollution policy, these ancillary benefits may look like a particularly good deal if Annex I countries pay for the GHG reductions in the first place.

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<sup>26</sup> The second method is in fact a special case of the use of income weights (see below).

<sup>27</sup> The use of average values for damages implies income weights based on an elasticity of social marginal utility of income ( $\epsilon$ ) of one. See also Fankhauser *et al* (1997).

The purpose of this paper has been to provide a conceptual framework for evaluating this position. We considered ancillary benefits in the context of standard welfare economic theory, examined various types of claimed ancillary benefits to determine the conditions under which these claims are valid, identified factors that could lead to both far lower and far higher ancillary benefits than have been claimed, examined the possibilities that economic behavior would bring ancillary costs rather than benefits, and paid special attention to these issues in a developing country context.

The results of our investigation generally work to constrain claims of very large ancillary benefits, although some of our results point to possibilities of far larger ancillary benefits and others are neutral, in the sense that they show how sensitive ancillary benefit estimates are to economic and policy conditions. To summarize the framework we begin to develop, we consider our findings in what we believe is the most important order for analysts to consider and include in studies, and for policy makers to use as criteria in evaluating the quality of ancillary benefit studies.

1. The estimation of ancillary benefits requires localized models of environmental impacts, population, exposure, preferences and valuation. The result is expected to be estimates that vary significantly by nation or region. Indeed, *methods should be consistent and estimates should not*. Furthermore, the estimation of ancillary benefits avoids many of the most vexing problems in economics and philosophy that characterize other aspects of climate change analysis. While in general, climate change problems transcend regions and generations, ancillary benefits largely accrue in the present and in a institutional context that is largely commensurate with policy making (the national level).
2. We find that economic behavior in response to climate policy-induced market signals can lead to ancillary costs. Input substitution in production, substitution to unmonitored fuels, movement of energy production and consumption to countries lying outside the climate policy regime (the ancillary leakage effect), and greater reliance on carbon sinks with their attendant environmental costs all can act to create ancillary costs. It is not possible to gauge how large these costs could be and whether they could fully offset ancillary benefits. The costs of a greater reliance on nuclear or hydroelectric power, diesel fuel for transportation, and the leakage of carbon emissions to developing countries, total to significant potential ancillary costs. However, we think a full offset of ancillary benefits at a global or at the national level is highly unlikely.
3. The size of ancillary benefits is directly tied to the baseline against which such benefits are to be estimated. If, following the environmental Kuznets' curve, one assumes that controls on conventional pollutants will be tightened over time and structural economic changes will lead to less pollution per unit output over time, then residual pollution or other externality-creating activities creating ancillary benefits will shrink over time. Given the extreme sensitivity of ancillary benefits to baseline assumptions, we mainly plea for transparency in assumptions between cost and ancillary benefit analyses and sensitivity analysis. In addition, if ancillary benefit estimates are to be compared directly with the costs of GHG policies, then the assumptions especially about the baseline need to be consistent in these estimates.
4. The size of some classes of ancillary benefits (the non-environmental categories) may be smaller than is commonly assumed because of the difference in the *existence* of an externality and its *relevance for policy purposes*. For example, traffic fatality reductions and workplace fatality reduction only count insofar as auto insurance does not internalize the costs to drivers and employer liability or normal labor market behavior does not internalize these costs to employers. As another example, the benefits of technological change brought about by the GHG policy could be counted if one argues convincingly that some market failure has prevented such technological change from being otherwise realized, i.e., *that it would have been realized had the market failure been absent*. In general economists are skeptical that such unexploited opportunities are pervasive.

The treatment of unemployment is another example, where any transitional increases in employment may count as ancillary benefits to the extent that labor markets fail. Here again, most economists are skeptical that these effects can be large in well-functioning economies.

5. A variety of our conclusions address the sensitivity of ancillary benefits to micro-level details, particularly spatial ones, but also details of policy. Because health and other effects depend on the spatial relationships between emissions sources and receptors, even with large reductions in GHG emissions, alteration in the spatial distribution of these emissions can result in larger or smaller than proportional increases in ancillary benefits. Regarding the effects of policy details, a particularly important one is whether the conventional pollutant is already being regulated with a policy internalizing externalities. In the case of an enforced cap on the pollutant, to the extent that a climate mitigation policy would result in cost savings in meeting the cap, these savings count as ancillary benefits. Burtraw et al (1999) show that such savings could be large (up to 30%) as a fraction of climate change mitigation costs, at least for initial bits of reduction in carbon emissions, for the electricity industry in the U.S., which is subject to an SO<sub>2</sub> cap.
6. Our discussions about the value of better information for conventional environment externalities point out how certain research advances may lead to larger ancillary benefits than are commonly estimated (mainly because so many potential external effects remain unquantified and unmonetized) and some may lead to smaller estimates. In the former category are the quantification and monetization of materials damages and marginal changes in ecosystem quality. In the latter category would be findings that would lower the mortality and chronic morbidity risks of air pollution and the preferences for reduced risks of these health endpoints.
7. Particular to developing countries, we find that the transfer of values from developed countries to developing countries to monetize ancillary effects is a problematic practice. The decisions to participate in climate policy are local to the country — meaning that both local costs and local ancillary benefits should be compared (abstracting for the moment about regional effects). Therefore, indigenous estimates of value are needed to properly assess whether the consideration of ancillary benefits should alter a country's climate policy positions or activities. Benefit transfers are clearly a far second-best approach to obtaining such values. In the absence of WTP estimates, it may be useful to “retreat to defensible border” by relying on estimates of benefits that provide a firm lower bound on the full measure.
8. In developing countries issues of employment and equity also play a bigger part than in developed countries. Projects and programs must be judged in this broader framework, giving due weight to these concerns. The tools for analyzing them have been discussed in this paper. They include measures to convert employment and equity effects into monetary units, so that they can be compared with other ancillary costs. But in many cases policy makers will simply want to know, in some detail, what these effects are.
9. Our results suggest that CGE modeling of climate change mitigation costs and changing patterns of energy consumption will need to be complemented by local scale models for estimating ancillary benefits and costs, perhaps in more of a case study approach, since such modeling on a global scale will be prohibitive.



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## APPENDIX: DEFINITIONS

One of the key issues is to ensure that the estimation of ancillary benefits and costs is done in a consistent manner and in the same conceptual framework as other costs and benefits. This section provides the definitions of some of the main terms that are used in the literature. An appendix provides some guidance on the basic issues that arise in the estimation of these benefits and costs.

### Opportunity Costs, WTP/WTA

The key concepts in the assessment of all benefits and costs are **social costs, private costs and externalities**. Underlying these are the notions of **opportunity cost, and willingness to pay/willingness to accept**. This section provides a non-technical review of these concepts, and a discussion of how their relevance to the ancillary cost/benefit debate.

The conceptual foundation of all cost estimation is the value of the scarce resources to **individuals**. Thus values are based on individual preferences, and the total value of any resource is the sum of the values of the different individuals involved in the use of the resource. This distinguishes this system of values from one based on 'expert' preferences, or on the preferences of political leaders. These values are measured in terms of the **willingness to pay (WTP) by individuals to receive the resource or by the willingness of individuals to accept payment (WTA) to part with the resource**. The costs of WTP and WTA therefore play a critical part in the whole cost methodology. A frequent criticism of this basis of costing is that it is inequitable, as they give greater weight to the 'well off'. While acknowledging the validity of this criticism it is important to note that there is no coherent and consistent method of valuation that can replace the existing one in its entirety. Where there is a concern about equity that should be addressed separately from that of cost estimation. The estimated costs are only one piece of information in the decision-making process for climate change.

In parallel with this, a second foundation of all valuation is the notion of **opportunity cost**. The opportunity cost of providing a commodity or service (call it X) is defined in terms of the value of the scarce resources that have been used in producing X. Those in turn are measured in terms of the value of the next best thing, which could have been produced with the same resources. This notion of cost may differ greatly from the common notion of cost. For example, take the cost of sequestering carbon by growing trees on a tract of public land. In estimating the costs of such a program, what do we take as the cost of the land? In some cases a zero 'cost' is attached, because the land is not rented out and no money actually flows from the project implementor to the owner. This, however, is incorrect in terms of opportunity cost. The cost of the land is to be measured in terms of the value of the output that would have been received from that land had it not been used for forestry. Such output may be a market good or service (e.g. agricultural output), and/or a non-market good or service (e.g. recreational use)<sup>28</sup>.

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In some cases recreation benefits may be marketed. Other examples of non-marketed services include soil erosion control and biodiversity conservation.

The two concepts of WTP/WTA and opportunity cost come together because opportunity cost is measured in terms of WTP/WTA. To make the example concrete, consider the example of hiring one day of labor by a construction company as part of the program of building a dyke. The WTA payment for that day of work will be equal to the value the worker attaches to the best alternative use of the time, which is the opportunity cost of that time to the worker. As for as the payment offered by the employer, the WTP will be no greater than the value of the alternative use to which the payment could be put. Hence both the WTA and WTP concepts are related to the concept of opportunity cost<sup>29</sup>.

## Social Costs and Benefits

In calculating the opportunity cost of producing a good or a service we must take account of the full opportunity cost, measured as the value of the best alternative use to which the all resources employed in producing the good or service could be put. Each of these alternative use values is in turn measured in terms of the WTP/WTA of the individuals who own the resources affected by the production process. If all the resources are accounted for in this way we have a cost that can be defined as the **social cost**.

Such a social cost may not be equal to the financial cost of a commodity or service. The financial cost of supplying electricity generated from a coal-fired power station will include payments to labor, capital and raw materials. This will not equal the social cost, however, if (a) the payments are not based on the opportunity costs of the labour, capital etc. and (b) resources such as clean air have been used up in the production of the electricity and payment is not made to those affected by the loss of that resource, based on its opportunity cost. The financial cost can also be referred to as the **private cost** of supplying the electricity if all resources under the control of the supplier are paid for in financial terms. If some resources (e.g. own labor) are not so paid for, the financial cost may differ from the private cost.

One of the most important reasons why the financial or private cost may differ from the social cost is the presence of **external effects or externalities**. Externalities are said to arise when the production or consumption of something has an impact on the welfare of someone, and that welfare effect has not been fully taken into account by the persons responsible for the production or consumption decision. In the above example, the welfare costs of air pollution from the generation of the electricity may not be taken into account by the suppliers of electricity. To fully take account of this welfare effect, the persons affected by the loss of air quality would have to agree to the loss based on their WTA payment.

The key points of note with regard to opportunity cost therefore are the following:

- The opportunity cost of a commodity is measured in terms of the value of the best alternative use to which resources used in making the commodity could be put. That is turn is given by the WTP/WTA for releasing the resource for its present use by the individuals who own the resources.

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<sup>29</sup>

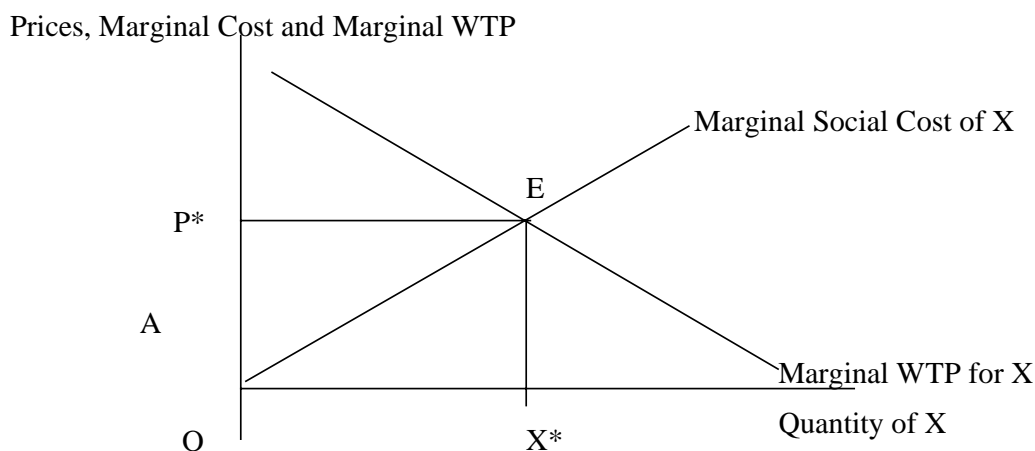
In a competitive market the WTA and WTP values are equal for the last worker hired. Where the WTA and WTP values differ, we need to choose between them. This issue is discussed further below.

- The social cost of producing a good or service is given by the opportunity cost of all the resources that go into producing it. Some of these may not involve financial payments. Hence the financial cost may not be equal to the social cost. The financial cost is equal to the private cost if all resources provided by the party responsible for the good or service are paid for in money.
- The financial cost or private can differ from the social cost for number of reasons. The most important of these is the presence of external effects. These arise when the welfare of individuals is affected by the production and/or consumption of something but full account is not taken of that effect.

### Market Prices, Marginal Private Costs, Marginal Social Costs and Externalities

In well functioning markets, where prices are determined by trades between many buyers and sellers, and where prices exist for all scarce resources, these prices are equal to the **marginal** social costs. By this we mean that the price gives the social cost of producing the last unit of the good or service. Figure A-1 below shows how the prices are determined and related to the marginal social cost and price.

Figure A-1. **Marginal social cost, marginal WTP and price**



In competitive markets producers supply goods to the point at which the price is equal to the marginal cost of production. If the latter includes all elements of cost it is called the marginal social cost curve and the market price is determined at the point at which the marginal social cost is equal to the demand for the good. The demand is in fact the marginal willingness to pay for the good or what the consumers are willing to pay for one more unit of X. Hence the equilibrium price P\* and quantity X\* are at a point where:

$$\text{Marginal Social Cost of X} = \text{Marginal Willingness to Pay for X} = P^*$$

Note also that the marginal social cost, by definition is also equal to the sum of the opportunity costs of the inputs used in producing the last unit of X. The total social cost of production is given by the area under the marginal cost curve, or OAEX\*.

In practice markets do not work as efficiently as Figure A-1 would suggest. In particular, it is important to allow for (a) the presence of externalities and (b) the possibility that private costs do not reflect opportunity costs because of other market and government failures.

### **Externalities, private cost and social cost**

Externalities also result in a deviation between marginal private costs and marginal social costs. In Figure A-2, a negative externality will imply that the marginal private cost is less than the social cost. With the lower private cost curve, suppliers will provide an amount of  $X$  at the point where the marginal private cost curve cuts the demand curve ( $X^{**}$ ). At this point the amount of  $X$  is greater than the socially desirable level. The difference between the marginal social cost and the marginal private cost is the marginal external cost. At output  $X^{**}$  this is given by the distance  $GF$ .

We should note that  $X^*$  is 'socially optimal' and  $X^{**}$  is not. The reason is that the social cost of producing at  $X^{**}$  is  $OAGX^{**}$ , an increase over that of producing  $X^*$  of  $X^*EGX^{**}$ . The consumers who receive the additional production of  $X^{**} - X^*$  value it at the area under the marginal willingness to pay curve, i.e. at  $X^*EFX^{**}$ . Hence in moving from  $X^{**}$  to  $X^*$  there is a net gain, the difference between the savings in social cost less the loss of the value to the consumers. In Figure A-2 this is equal to  $X^*EGX^{**} - X^*EFX^{**}$ , or  $EGF$ . It can be shown that any point other than  $X^*$  will generate some net gain when a move back to  $X^*$  is considered.

### **What are externalities and why do they arise?**

Externalities arise when there are incomplete markets. If the production of  $X$  requires a resource for which there is no market, the producer of  $X$  will use that resource without taking account of the opportunity cost of the affected party, or equivalently of that party's WTA payment for the amount of the resource that is used. Hence the marginal cost of supply does not equal the marginal social cost<sup>30</sup>. Such incomplete markets can arise for several reasons. One is insufficiently defined property rights. Another is the indivisibility of the resource; for example clean air cannot be 'owned' by a single person. Even if the 'right' to clean air is vested with the residents of a locality they cannot individually sell off their rights. An extension of this indivisibility occurs when the resource is a global one, such as the stratospheric ozone layer.

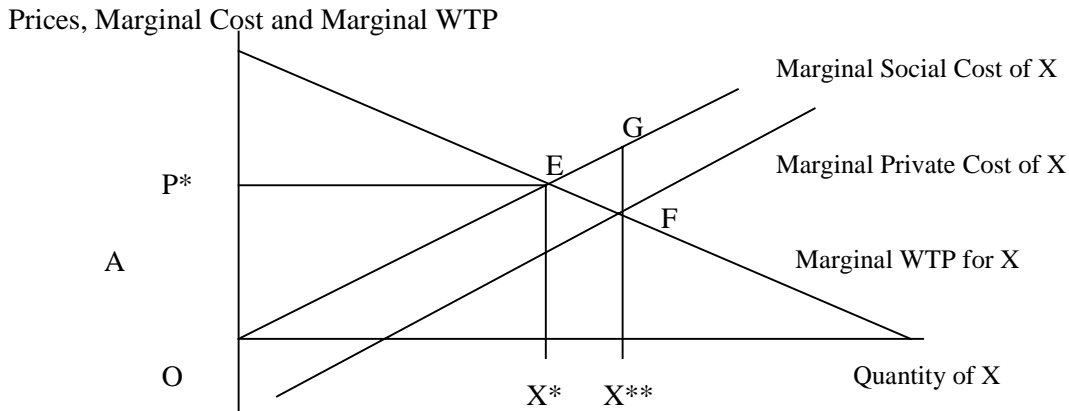
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In some cases suppliers may in fact behave as if they were facing the marginal social cost curve, if bargaining between producers and affected parties is possible. This is the famous Coase (1960) result but it is unlikely that such bargaining can take place in the context of most of the ancillary benefits of interest here.



Figure A-2. **Externalities, Marginal Social Cost, Marginal WTP and Price**



Examples of externalities that are relevant to the ancillary benefit debate include the following:

- Changes in pollutant emissions from the burning of fossil fuels.
- Increase in the availability of forested areas for recreational use as a result of reforestation.
- Reductions in public property losses from fires when lighting and heating by kerosene is replaced by electricity.

### **Measuring Externalities**

The analysis in Figure A-2 assumes that the externalities can be measured in money terms, using values based on opportunity costs. The process of obtaining money values requires interdisciplinary work, with scientists and engineers providing data on the physical impacts and economists valuing these impacts. An example is given by the ‘impact pathway approach’, for air pollutants, where emissions of pollutants are measured and their dispersion and chemical interactions modeled, to provide information on the spatial distribution of concentrations. The concentrations are used to estimate the physical impacts - on health, materials, crops, ecosystems etc. Finally values are attached to the impacts, based on WTP/WTA, and models of human response to the impacts. Examples of such analysis are to be found in Lee (1995), ExternE (1999). As those studies show, however, monetization is not always possible and certainly not for all impacts. Some physical effects, particularly on water bodies and natural and semi-natural ecosystems, can only be reported in physical terms. In such cases the characterization of the externality has, necessarily, to be partly in money terms and partly in physical terms.

### **Issues in the definition and estimation of externalities**

There are a number of complications that arise in applying the theory of externalities, which anyone working in the field has to be aware of. These are:

- The distinction between market externalities and pecuniary externalities.
- The importance of the regulatory framework in assessing the policy relevance of measured externalities.
- The need to be consistent in the accounting framework.

### *The distinction between ‘real’ externalities and pecuniary externalities*

The externalities described above are ‘real’ in that they result from a lack of markets and are relevant in comparing private and social costs. The literature also refers to pecuniary externalities, or changes in incomes resulting from a GHG mitigation project. Such changes can be thought of as ‘externalities’ in the sense that the impact on the person or firm is unexpected and/or unintended. But they are not to be confused with the market externalities. The following examples should help clarify the picture.

A carbon sink project creates an improvement in a local amenity by reforestation. The market externality is measured in terms of the WTP of the individuals who would use the amenity. This use could, however, also generate an increase in the fees received by the park owners, and increased profits by facilities in the region that provide services - restaurants, hotels etc. The latter are all pecuniary externalities and including them in the assessment would amount to double counting.

A GHG mitigation transport project reduces air pollution by switching from private to public transport. One of the impacts of this switch is the increase in income for the providers of public transport and the fall in income for car maintenance services, car sales outlets etc. These changes operate through organized markets and are not relevant to the estimation of the external effects of the reduction in air pollution.

### *The importance of the regulatory framework in assessing the policy relevance of measured externalities*

When external effects are present, governments may take actions to mitigate them. These actions may reduce the gap between the ‘socially optimal’ level of the good and the amount that the private sector would supply. In terms of Figure A-2, for example, a tax on the private supplier equal to FG per unit of X would result in the private supplier having a marginal private cost curve that results in an output decision of X\*. In that event an externality is still present, but it is no longer ‘relevant’, in the sense that no further adjustments to output need to be made on account of the externality.

This is an important point because it underscores the distinction between the **existence** of a measured externality using methods such as those described above, and the **relevance** of the externality for policy purposes. The latter requires much more information - especially about the role of the regulations in bringing the social and private decisions closer together. The issue arises in the context of ancillary benefit estimation and is discussed further in Section IV.

### ***The need to be consistent in the accounting framework***

There are some areas where the theory of externalities applies less clearly than others. The above examples of ‘real’ externalities are ones where hardly anyone would doubt the relevance of the theory. But there are also some less clear-cut cases. One is the in the change in efficiency in the use of technology brought about by GHG mitigation measures. The problem here is the definition of the ‘market’, which is missing and the fact that the efficiency gains occur to individuals who should be able to take account of such gains in their decision-making. The case for treating such gains as external is that individuals may not be aware of the possible gain and therefore the ‘missing market’ is that for information on the effects of adopting different technologies on future costs of those technologies. What is decided as to the externality status of this impact is, therefore, a matter of debate. **If**, however, we treat technological efficiency as an externality, it is critical that the opportunity costs of obtaining the improvements in technology, including R&D elements are included as cost items.

A second example of an ‘externality’ where there is some doubt is with respect to reductions in fatalities from road accidents when there is a shift from private to public transport, or the reductions in death rates from reductions in air pollution. In both cases individuals acting rationally should be aware of such impacts and, in principle, the property rights for safety and clean indoor air are well defined. The issue, as in the case of technology efficiency is information. If individuals are not aware of the risks, or are not able to exercise a choice with respect to the risk level they would like because of market imperfections, there is case for including them as externalities. In that event, however, it is important to include all costs associated with the change in behavior resulting from the programme or policy that reduces these risks.

### **Other government and market failures**

The discussion so far has been about externalities, or situations where missing markets cause a divergence between private and social costs. Such divergences can also arise, however, for other reasons. Prominent among them are:

- Government subsidies and taxes.
- Government controls that restrict, in one form or another, the supply for demand for a particular input or output.
- Market imperfections such as monopoly or monopsony power.

All these factors result in market prices deviating from marginal social costs. Hence in making a proper assessment of the social cost this divergence has to be allowed for. The following are some important examples of divergences and how they may be addressed.

## Labor and capital market imperfections

Adjustments to for deviations between private costs and opportunity costs in labour and capital markets are made through the use of ‘shadow prices’. As an example, if the wage paid to a worker is \$30/day and the opportunity cost of his or her time is only \$15 a day, a shadow price of 0.5 is applied to the actual wage to get the social cost of that input. The underlying imperfection in this case may be union power which keeps the wage artificially high, or macroeconomic failure, which prevents the labour market from clearing. Likewise, where capital markets are distorted, the market price of capital may not reflect its true scarcity. This would imply the need to apply a shadow price to capital of greater than one; something which is routinely done in project appraisal by public sector investors. The details of how such premia can be calculated are discussed in standard treatments of the subject (see, for example, Ray (1984)).

## Shadow Prices for Goods and Services

The standard assumption for social cost pricing of all goods and services is to take the international prices<sup>31</sup> for all tradable commodities. This assumes that international prices are indeed free market prices. If that is not the case, an adjustment must be made to the international price to reflect any divergence from social cost. Note that such adjustments will mean that all inputs and outputs will be valued net of any taxes or subsidies.

## The discount rate

The debate on discount rates is a long-standing one. As the SAR report notes, there are two approaches to discounting; an ethical approach based on what rates of discount **should** be applied, and a **descriptive** approach based on what rates of discount people (savers as well as investors) actually apply in their day-to-day decisions. The former leads to relatively low rates of discount (around 2-3% in real terms) and the latter to relatively higher rates (at least 5-6% and, in some cases, very much higher rates).

Although there is a good case for using low discount rates to evaluate climate change impacts (see, for example, the discussion in the SAR), the same does not apply to mitigation programs and policies. For these, the country must base its decisions at least partly on discount rates that reflect the opportunity cost of capital. In developed countries the rates are around 4-6% would probably be justified. Rates of this level are in fact used for the appraisal of public sector projects in the EU (Watts, 1999). In developing countries the rate could be as high as 10-12%. The international development banks use these rates, for example in appraising investment projects in developing countries. It is more of a challenge, therefore, to argue that climate change mitigation projects should face different rates, unless the mitigation project is of very long duration.

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<sup>31</sup> The relevant prices are the international price f.o.b. from the country concerned for goods that are exports or potential exports, and c.i.f. to the country for goods that are imports or import substitutes. Where goods are not tradable, shadow prices are estimated using the costs of production with inputs priced at international prices.

In addition to discounting future costs and benefits of climate change and mitigation programs, there is the further issue of whether or not future emission reductions should be discounted when compared to present reductions. The justification for discounting them is that future reductions are worth less than present reductions in terms of reduced impacts. The choice of the appropriate rate, however, remains an unresolved issue. A recent survey of discount rates applied to carbon flows reveals values ranging from 0 to 10 % (Boscolo, Vincent and Panayotou, 1998). Some studies do not apply a discount rate but simply take the average amount of carbon stored over the project life-time (referred to as flow summation) or take the amount of carbon stored per year (flow summation divided by the number of years). Both these methods are inferior to applying a discount rate to allow for the greater benefit of present sequestration over future sequestration. The actual value, however, remains a matter of disagreement, but the case for anything more than a very low rate is hard to make.

## **BASELINE ISSUES IN THE ESTIMATION OF THE ANCILLARY BENEFITS OF GREENHOUSE GAS MITIGATION POLICIES**

by Richard D. MORGENSTERN

### **Abstract**

*Greenhouse gas reduction policies which alter fossil fuel use can have near term environmental and social impacts quite distinct from the longer term benefits for climate change for which they were originally designed. The air pollution-related health improvements that accompany the reduction in GHGs are the best understood of these so-called ancillary or co-control benefits. Impacts on traffic congestion, ecosystem health, safety, and others are also potentially important, although to date they have been less well studied. Current estimates of the monetized health impacts associated with reductions in the use of carbon-intensive fuels range from \$3 to several hundred dollars per ton of carbon abated. Reductions in the costs of meeting existing pollution control requirements – so called avoided costs - can add to these benefits.*

*Some experts have questioned whether estimates of ancillary benefits are largely an artifact of the (unrealistic) assumptions used to generate them. It is argued, for example, that previously established environmental policies, technological, demographic, economic or other patterns already underway will improve those areas that some would count as ancillary to GHG reduction policies. If that were the case then there could be a significant problem of double counting in the estimation of ancillary benefits.*

*Without a credible and consistent specification of the health, environmental, economic or other conditions that occur in the absence of the contemplated policies it is not possible to estimate the true impact of such policies. To date, few studies have specified how the health, environmental or other ancillary benefits are expected to deviate from their current levels in the absence of GHG mitigation policies. Explicit and transparent specifications of the baseline conditions relevant to policy, technology, demography and other factors are crucial to the estimation of ancillary benefits.*

*This five-part paper catalogues and analyzes the broad set of baseline issues that must be addressed in order to conduct an informed policy debate on ancillary benefits. Following the Introductory Section, Section II provides background on some of the major types of ancillary benefits and reviews the literature on the health studies of air pollution. Section III develops a framework for considering baseline issues and identifies the critical baseline issues posed by current studies. Section IV provides a prescriptive checklist of key baseline issues. Section V offers a series of broad conclusions and, in the context of recent IPCC discussions, suggests that the expanded use of carefully done case studies offers the best means to estimate ancillary benefits at this time. Throughout the paper efforts are made to balance analytic robustness against the practicalities of policy analysis. As much needed information is missing and many data gaps exist, this paper also provides some preliminary guidance with respect to research priorities that need to be addressed in order to improve the ability of the research community to assess the full range of ancillary benefits of GHG mitigation policies.*

## 1. Introduction

It is now well established that greenhouse gas reduction policies, which create incentives to alter the uses of fossil fuels, can have near term environmental and social impacts quite distinct from the longer term benefits directly associated with climate change. The air pollution-related health improvements that accompany the reduction in GHGs are the best understood of these so-called ancillary or co-control benefits. Impacts on traffic congestion, ecosystem health, safety, and others are also potentially important, although to date they have been less well studied.

Current estimates of the monetized health impacts associated with reductions in the use of carbon intensive fuels range from \$3 to several hundred dollars per ton of carbon abated. Reductions in the costs of meeting existing pollution control requirements – so called avoided costs - can add to these benefits. Even the low and mid-range estimates could offset a significant portion of the projected GHG abatement costs and thereby add new meaning to the interpretation of ‘no regrets’ policies. Inadequate consideration of ancillary benefits could lead to the selection of inappropriate GHG mitigation policies because of the failure to capture the full range of benefits. However, many problems remain in developing credible numbers.

Some experts have questioned whether estimates of ancillary benefits are largely an artifact of the (unrealistic) assumptions used to generate them. It is argued, for example, that previously established environmental policies, technological, demographic, economic or other patterns already underway will improve those areas ancillary to the GHG reduction policies. If that were the case then there could be a significant problem of double counting in the estimation of ancillary benefits.

It is well known that without a credible and consistent specification of the health, environmental, economic or other conditions that occur in the absence of the contemplated policies it is not possible to estimate the true impact of such policies. For example, the baseline scenarios for greenhouse gases over the next one hundred years have been widely debated by the Intergovernmental Panel on Climate Change (IPCC, 1998). Strong views have been expressed that the draft IPCC baseline scenarios do not reflect the full range of plausible GHG emission futures. Errors in these baselines can bias estimates of the direct benefits and costs of GHG mitigation policies.

An important but less well-explored area concerns the baselines used for a host of ancillary benefits. To date, few ancillary benefit studies have developed counterfactual scenarios explicitly tailored to the ancillary issues. That is, they generally do not specify how the health, environmental or other ancillary benefits are expected to deviate from their current levels in the absence of GHG mitigation policies. For the same reason that understanding the baseline emissions of GHGs is essential to estimating the direct benefits and costs of mitigation, explicit and transparent specifications of the baseline conditions relevant to policy, technology, demography and other factors are crucial to the estimation of ancillary benefits.

The purpose of this paper is to catalogue and analyze the broad set of baseline issues that must be addressed in order to conduct an informed policy debate on ancillary benefits. Section II provides background on some of the major types of ancillary benefits and reviews the literature on the health studies of air pollution. The review includes papers from the U.S., Europe, and to a very limited extent, developing countries and economies in transition. While a number of the papers have been published in the peer-reviewed literature, some are sufficiently new that they are still circulating in draft form. Section III develops a specific framework for considering baseline issues and identifies the types of baseline issues most likely to affect ancillary benefit estimates. Section IV provides a prescriptive checklist of key baseline issues. Section V presents the overall conclusions. Throughout this paper efforts are made to balance analytic robustness against the practicalities of policy analysis. As much needed information is missing and many data gaps exist, this paper also provides some

preliminary guidance with respect to research priorities that need to be addressed in order to improve the ability of the research community to assess the full range of ancillary benefits of GHG mitigation policies.

## 2. The Nature and Types of Ancillary Benefits

### 2.1 Definitions

To shed light on the conceptual issues we begin by asking what do we mean by “ancillary benefit” of a GHG reduction policy? Stated simply, ancillary benefits (also known as secondary or co-control benefits) are those which accrue as a side effect of policies targeted at a particular problem. If such benefits legitimately depend on GHG reduction policies then they should be considered in the overall costs and benefits of the GHG policies. The inclusion of ancillary benefits in the overall mix of benefits associated with GHG reduction policies can affect both the optimal level of abatement as well as the particular policy instruments selected.

Not all ancillary benefits are positive. Negative ancillary benefits indicate a conflict between or among policy objectives. Particularly large negative ancillary benefits could even transform a worthwhile policy into one that is not worthwhile. Conflicts among policy objectives exist in a number of well-known areas relevant to GHG mitigation. For example, diesel vehicles (lower GHG, higher particulate emissions), natural gas fire co-generation in urban areas (lower GHG, higher urban  $\text{NO}_x$ ), or non carbon based technologies like nuclear power or wind turbines (lower GHGs, but increases in other risks) all present particular risk-risk trade-offs. While none of these negative effects have been formally quantified in the ancillary benefits literature *per se*, the trade-offs they represent have been extensively studied elsewhere.

To determine the ancillary benefits (or costs) of a GHG policy, one must compare conditions in a world with the policy to conditions in a world without it. To produce such estimates, both the “with” and the “without” scenarios must be modeled; they cannot be observed. The quality of the information available to develop estimates of these scenarios is limited by three basic uncertainties:

- i. What economic actors (firms, households) in a particular country are doing at the present time with respect to air, water, waste, emissions, accidents, etc *independent of* any proposed GHG mitigation policies?
- ii. What these actors will do *in response to* the new GHG policies?
- iii. What these actors *would have done* in the future if the GHG policies had not been adopted?

The first of these items is, in principle, knowable *ex ante* but in practice is often not fully understood by policy makers. The second and third items are hypotheticals, based on economic and process-analysis models and, perhaps, qualitative information from industry or other experts.

Interestingly, even after implementing a policy it is not easy to determine the true benefits (costs) of the policy, since the world with the policy is observed, but the counterfactual is not. The *ex post* cost estimate must deal with the same uncertain elements but from a more favorable position, especially on items (i) and (ii). It can be no worse than the *ex ante* estimate, because it has more information to draw on. What is still missing is information on item (iii), the counterfactual, although better information is usually available *ex post*. Thus, one observation is that the definition of ancillary benefits, like that of any other type of benefit (or cost), is somewhat arbitrary, depending as it does on the analysts’ beliefs on what would have happened without the policy.



## 2.2 Categories

Acknowledging the somewhat arbitrary nature of estimates of ancillary benefits, however, does not diminish the importance of developing credible and consistent baselines. The most commonly discussed ancillary benefits are: health, ecological, economic/welfare, and safety. Other benefit categories, e.g., employment or technological change (both of which could lead to increases in GDP growth) or what Pearce (2000) refers to as community severance (loss of community due to increased traffic flows) may also be relevant.

### 2.2.1 Health

Health benefits, including both morbidity and mortality, are the most studied and represent, by far, the largest category of estimated ancillary benefits. They are considered in some detail in a later part of this section. The issues relevant to baselines *per se* are discussed in Section 3.

### 2.2.2 Ecological

Some experts believe that ancillary ecological benefits, though largely unstudied, may well be an important category of ancillary benefits. The deposition of air pollutants, including nitrogen compounds, is a potentially important category of ancillary ecological benefits. Reduced water discharge or changes in runoff and soil erosion are other possible categories. A paper by Krupnick *et al.* (1998) finds that airborne NO<sub>x</sub> emission reductions slated to occur under the 1990 Clean Air Act significantly reduce nitrate loadings to the Chesapeake Bay. Although this study did not monetize the benefits of these reductions nor did it specifically tie them to carbon reduction policies, other work by some of the same authors has estimated ancillary health benefits associated with reduced NO<sub>x</sub> emissions (Burtraw *et al.* (1999)). A paper by Aunan *et al.* (1998) suggests that forests in large parts of Europe are probably adversely affected by air pollution although, as they note, “the understanding of the causes and mechanisms is poor except in the most polluted areas where direct effects are plausible.” It is thus reasonable, although not yet specifically modeled, to presume that ecological benefits via reduced airborne emissions may be a significant source of ancillary benefits.

A modeling effort recently established in Europe is beginning to look beyond airborne emissions and focus on direct water discharges associated with GHG policies (RIVM *et al.* 2000). Various types of both user and non-user benefits are plausible on both the air and the water side although, as indicated, they have not yet been specifically modeled as ancillary benefits of GHG reduction policies.

### 2.2.3 Economic/Welfare

Economic or welfare benefits are another potentially important category of ancillary benefits. To develop credible estimates of benefits in this area it is sometimes important to distinguish between the impacts on stocks vs flows as part of the methodological framework.

A paper by Barker (1993) examined the relationship between the proposed EU Carbon/Energy tax and a number of economic/welfare categories associated with transportation including road surface maintenance expenditures, traffic noise, and congestion. He estimated ancillary benefits in these categories associated with the proposed EU tax amounting to about .05 per cent of 1990 GDP. He noted that his estimates are likely to underestimate the total economic/welfare ancillary benefits from the categories examined because his model failed to capture a potentially important feedback namely, the reduction in air emissions associated with the resulting higher average driving speeds. Thus, a full

specification of the transportation-related ancillary benefits would likely include some further health improvements. In a similar vein, a recent Australian paper examined how GHG mitigation policies including road pricing would reduce traffic congestion (Australian Bureau of Transport and Communication Economics, 1996).

Aunan *et al.* (1998), in their study of Hungary, explore the relationship between energy saving programs and reductions in materials damage. They also examine potential increases in crop yields associated with reductions in air pollution (ozone). The materials damage study is based on a detailed analysis of building mass and materials in Budapest, together with results from other studies in Europe. They find that the 6 per cent reduction in SO<sub>2</sub> concentrations associated with the implementation of the energy saving programs leads to annual reductions in materials damage on the order of \$30-35 million. As the authors note, however, the fact that SO<sub>2</sub> concentrations have declined over the past few years implies that their baseline assumptions (constant 1990 levels) are too pessimistic which, in turn, suggests that their damage estimates are likely overstated.

The Aunan *et al.* (1998) analysis of the relation between Hungary's energy saving programs and increased crop yields is more preliminary in nature. It is based on an estimated linear relationship between the yield of wheat and ozone concentrations, and on (the quite limited) information on the local atmospheric chemistry of ozone formation. They find that energy saving programs in Hungary are likely to have only modest effects on crop yields in that country. However, they suggest that significant increases in yields are likely to be obtained if NO<sub>x</sub> and VOC emissions are reduced in large regions in Europe.

Gielen and Peters (1999) examined the effects of different levels of carbon taxation in Europe on waste management. Although they find little effect on total waste volume, they estimate significant changes in the mix of wastes. For example, they predict an increase in the share of paper and wood products. Further work needs to be done to understand the full economic implications of these effects on waste management (both positive and negative). Other candidate economic/welfare issues include the improved visibility and reduced materials damage likely to flow from GHG mitigation policies.

#### 2.2.4 *Safety*

Safety represents a further area of interest for potential ancillary benefits. In the same paper in which he examined the economic/welfare ancillary benefits, Barker (1993) modeled the reduced traffic accidents associated with the proposed EU carbon/energy tax. Not surprisingly, he found that even a small tax increase would lead to a significant reduction in fatal and non fatal traffic accidents. In the area of industrial safety one can certainly imagine that the broadscale economic effects associated with energy price changes could lead to output changes which, in turn, altered overall accident rates (e.g., substitution computer software for steel production), although the net effects on accidents are not clear. The Barker study is the only one we are aware of that has explicitly quantified the link between carbon mitigation policies and safety.

#### 2.2.5 *Other*

Depending how broadly one defines ancillary benefits, it is conceivable that other categories of ancillary benefits are also relevant. There is a well-known literature indicating that the move to performance-based regulation and away from technology-specific approaches can enhance innovation. To the extent that performance-based GHG mitigation policies substitute for technology-specific regulation, overall innovation may be encouraged. There is also a small but growing literature specifically focused on induced technological change, i.e., how much additional economy-wide

innovation, if any, would be stimulated by GHG mitigation policies. Unfortunately, there is no consensus of views in this evolving field. (For differing approaches see Grubb et al, 1995; Goulder and Schneider, 1999; and Goulder and Mathai, 1998) If one believed that GHG mitigation policies truly accelerated the overall rate of technological change then one might want to include ‘GDP growth’ as an explicit ancillary benefit.

### 2.3 *Avoided costs*

The most common way to think about categories of ancillary benefits associated with GHG mitigation policies is in terms of the direct effects on health, ecology, safety or other endpoints. However, depending on the types of policy in place GHG mitigation policies can also affect the cost of achieving particular policy goals. Burtraw and Toman (1997), EPA (1999), and Burtraw *et al.* (1999) all estimate abatement cost savings for SO<sub>2</sub> reductions under the allowance trading program as an ancillary benefit of GHG reduction policies. The basic idea is quite simple: as long as the SO<sub>2</sub> cap is binding, policies to reduce GHG emissions from the power and industrial sectors will not lead to further reductions of SO<sub>2</sub> emissions. However, significant abatement cost savings may accrue to those purchasing or otherwise acquiring SO<sub>2</sub> permits freed up by the GHG policy’s induced SO<sub>2</sub> reductions. For example, the EPA study examined alternative levels of GHG mitigation to assess their impacts on other pollutants in the year 2010. The study found that under a business-as-usual scenario 82 GW of sulphur scrubbers would be installed. With the modeled GHG policies in place, fuel switching reduces the scrubbers required to 63 GW for an annual saving of almost \$500 million. Interestingly, there may be a tendency to generalize this result to policy contexts where the cap is softer than in the SO<sub>2</sub> trading program. Arguably, Lutter and Shogren (1999) have assumed that the proposed ambient fine particle standard, particularly as it applies to the Los Angeles basin, is tantamount to a hard cap against which one can credit cost savings arising from GHG mitigation policies. For a variety of reasons discussed in a later section, we believe the attribution of large cost offsets in this instance is not appropriate.

### 2.4 *Adaptation*

Although the literature has focused on the ancillary benefits associated with mitigation policies, there may be significant ancillary benefits associated with adaptation policies as well. For example, actions to adapt to the effects of climate change, ranging from individual decisions to turn on air conditioners in the summer, to more aggressive (communal) actions to build sea walls or corridors for wildlife to migrate, could have implications for air pollution (probably adverse) or ecological health (potentially positive).

A recent presentation by Scheraga (1999) highlights the potential for ancillary benefits from adaptive responses that could be taken to address climate change. Egypt, for example, has a number of populous and low lying areas vulnerable to sea level rise and thus threatened by salt water intrusion. Water resources in the Nile River Basin would be threatened with resulting risks to several important sectors. Agriculture, which would be affected directly by temperature changes, would also be indirectly affected by changes in the availability of water from the Nile. Heat stress would directly threaten human health. Various adaptation options exist to reduce these vulnerabilities. Improving irrigation efficiency, for example, would yield benefits to agriculture and, potentially, to human health. Scheraga points out, however, that such a policy would also reduce the demand for Nile River water, thus yielding benefits (or cost offsets) for the supply of drinking water, for hydropower requirements, etc. To our knowledge the ancillary benefits of adaptation policies have not been empirically examined, although this may be a fruitful area of research. [See also Strzepek, *et al.* (1995)].

## 2.5 *Air pollution and health*

The type of ancillary benefits most intensively studied in previous research involve the health effects associated with reductions in criteria (conventional) pollutants. A number of authors have recently reviewed the literature on this issue (Ekins 1996; Burtraw and Toman 1997; Burtraw *et al.* 1999; Pearce 2000; RIVM 2000). In this section we consider the same literature, by updating (slightly) previous reviews, and by including the handful of ancillary benefit studies conducted in developing countries. Table 1 enumerates the recent studies, along with a listing of the key methodological issues associated with each of them.

The most robust finding in this literature is that there are, indeed, significant ancillary benefits arising from the reductions in conventional pollutants associated with GHG mitigation policies, although the results vary considerably according to the countries and sectors studied, the nature of the policies examined, and other factors. This basic finding about the existence of potentially significant ancillary benefits holds up across every study we are aware of, which includes analyses conducted in the US, Europe, and a limited number of developing countries and economies in transition. In many of the studies the results are driven by reductions in NO<sub>x</sub> and CO. Depending on the country, smaller and generally less important contributions come from reductions in VOCs and/or Pb. Direct particle emissions (TSP or PM<sub>10</sub>) factor into a number of studies. However, secondarily formed compounds, e.g., sulfates and nitric acid, are not treated in a consistent manner in the literature. Thus, a potentially major source of elevated particulate concentrations – with potentially large health effects – is excluded from most of the studies.

Another finding from the literature is that the estimated ancillary benefits are considerably higher in Europe and in the limited literature on developing countries than in the US. Certainly the findings for developing countries are not surprising, given the higher baseline emissions levels for most conventional pollutants, the lack of in-place or planned standards, and the more aggregate level of modeling typically used in developing country studies.

The explanation for the differences between US and European studies, however, is a bit more complicated. Ekins (1996) reviews the (heretofore largely unpublished) European literature and indicates a best estimate of \$273 (1996 dollars) of ancillary benefits per ton of carbon reduced. The studies underlying this estimate include the initial fixed-coefficient papers plus a more recent series of analyses based on macro models [Barker *et al.* (1993), Alfsen *et al.* (1995)]. The economic valuation underlying these calculations are drawn from an early literature with values that are higher than those currently used in the US. (For current values used by the US Environmental Protection Agency, see EPA (1997)). Unfortunately, the valuation literature reviewed for the ExterneE Project – Europe's comprehensive fuel-cycle model of environmental impacts of new (1995) vintage power plants - was not incorporated into the Elkins review. Use of the ExterneE values, which are more in line with the lower US numbers, would have reduced the European ancillary benefit estimates considerably. Apart from these valuation issues, and differences in baseline assumptions (discussed below), the discrepancies between the US and the European results are probably attributable, as Burtraw *et al.* (1999) note to several other factors: a) the more aggregate level of modeling in the European studies, b) greater population density in Europe and c) the fact that a greater proportion of the US emissions are deposited offshore rather than on-shore as in Europe.

A corollary finding from this literature is that the estimates of ancillary benefits have declined over time, at least as regards studies within a single country or area. With the exception of the Lutter/Shogren study (discussed below) this is most evident in the US estimates, where some of the early studies derived estimates as high as \$80/ton of carbon abated (not shown in Table 1). More recent estimates are in the range of \$3-6 per ton of carbon abated, based on policy simulations involving modest GHG reduction policies (e.g., \$10/ton carbon tax). A similar story holds for the

European studies although, as noted, the estimates are generally higher and the secular decline is not as dramatic.

An equally important finding from the literature is that the differences in the valuation of the ancillary benefits – which vary by an order of magnitude – stem not only from the different policies and sectors studied, but from the great divergence of methods and models used in the analyses. The earlier studies typically employed a fixed coefficient modeling approach for estimating ancillary benefits (e.g., Pearce 1993). They attempted to calculate an average relationship between the reduction in carbon emissions and the reduction in conventional pollutants. While this served as a useful exercise in bringing to light the basic concept of ancillary benefits, it is now widely recognized that ancillary emissions reductions have no necessarily absolute or proportional relationship to carbon reductions. In fact, they can vary in complex ways depending on the nature of the GHG policies themselves, spatial location, and a host of other factors. For example, GHG-reduction policies that raise gasoline prices by a small amount may reduce driving somewhat and may result in ancillary emissions reductions roughly proportional to GHG reductions. However, larger changes in gasoline prices may result in changes in the fuel and vehicle mix that result in disproportionately larger reductions in ancillary emissions.

The early fixed-coefficient approach has now been supplanted by simulation studies which examine the effect of particular GHG reduction policies on nonGHG emissions in the context of a specific model. The clear advantage of the simulation approach is that it endeavors to capture more of the underlying complexity of the relationship between GHG policies and ancillary benefits. For example, in their work on electric utility models, Burtraw *et al.* (1999) find that the incremental value of ancillary benefits from NO<sub>x</sub> reductions per ton of carbon reduced declines with more aggressive carbon policies, although that calculation does not include the additional SO<sub>2</sub> reductions that may also accrue once SO<sub>2</sub> emissions fall below the legally mandated cap.

As Burtraw *et al.* (1999) note, the US simulation models are generally of two varieties, reflecting differences in approaches to estimating emission changes. In the first category are those studies which have linked economy-wide CGE models with estimates of emission rates in various industries to relate increases in the price of energy (generally via carbon taxes) to changes in investment and output and, in turn, to changes in CO<sub>2</sub> emissions, and in criteria air pollutants. The second category consists of studies, which use disaggregated models of the electric utility industry to look at changes in investment, utility operations, and conventional pollutant emissions associated with institutional and pricing reforms in the utility industry. In addition to the introduction of carbon taxes, examples of the policy changes studied in the disaggregated models include Green Lights, Motor Challenge and other voluntary programs embodied in the 1993 Clinton Administration Climate Change, and reform of electricity transmission pricing.

A fifth and critically important observation from this literature concerns the importance of the geographic location of the emission reductions. In the early 1990's major analytic efforts – carried out on both sides of the Atlantic – focused on the comprehensive effects of electricity fuel cycles (Lee *et al.* 1995; EC, 1995; Rowe *et al.*, 1995). One of the major conclusions of these efforts is that the environmental impacts and, more specifically, the monetized value of those impacts, depend critically on the geographic location of the emission changes. For example, Lee *et al.* (1995) estimate that the monetized value of the health impacts of a new coal-fired power plant are an order of magnitude higher for a plant located in Tennessee than in the less-densely populated state of New Mexico. Similar findings arise from all three of the cited fuel cycle studies. Other factors, including the potential for a nonlinear relationship between emissions and pollutant concentrations, or between concentrations and health effects, further enhance the importance of the complex, location-specific models.

Unfortunately, the aggregate CGE models cannot generate outcomes at a spatially-relevant level. The international CGE models typically divide the world into large regions (e.g., North America is typically considered to be one region). National models of the US treat the entire nation as a single area, although details are often available at the industry level. Thus, by their very design, the large CGE models are unable to capture the important locational differences that effect the valuation of emission reductions.

In contrast, the disaggregated models have the capacity to generate geographic-specific results. For example, the Holmes *et al.* (1995) study developed emission estimates on a geographic basis, according to the North American Reliability Council (NERC) Regions. The study also estimated emissions changes according to season and time-of-day, both of which would be important for any analysis of changes in ozone concentrations. Using the Holmes *et al.* emission estimates, Burtraw and Toman (1997) conducted a geographic analysis of atmospheric transport of pollution and the corresponding population exposure via the PREMIERE model. They estimated the benefits of the calculated NO<sub>x</sub> reductions at \$3.22 per ton of carbon reduced, roughly the same result obtained with the HAIKU model. (Burtraw *et al.* 1999).

Table 1. Recent ancillary benefits studies\*

Study	Spatial Area	Sector	Policy Assumption	Average Ancillary Benefits \$tC (1996 dollars)	Pollution Cost Savings Included	Comment
Lutter and Shogren, 1999	LA county	All	1990 Clean Air Act	\$300.	Yes	Integration of models and external calculation
Burtraw, <i>et al.</i> 1999	US	Electric	1990 Clean Air Act plus 1999 SIP Call	\$3. (for \$10 carbon tax)	Yes	Focus on NO <sub>x</sub> and SO <sub>2</sub> . Visibility and ozone benefits not considered
Cifuentes <i>et al.</i> 1999	Chile	All	Current policies explicitly considered (including 1997 Plan)	\$20-70	No	Six air pollutant categories. Detailed energy categories. Results sensitive to benefits transfer.
Dessus and O'Conner, 1999	Chile	All	Current policies explicitly considered, especially for PM <sub>10</sub> and Pb	\$150-300 (1992 \$ and exchange rates)	No	Seven air pollutant categories, including Pb. Results sensitive to benefits transfer methods.
Aunan <i>et al.</i> 1998	Hungary	All	1990-1992 policies assumed in place, including Pb in gasoline reductions.	NA	No	Bottom-up approach. Limited economic modeling. Multiple pollutants, endpoints and types of ancillary endpoints considered. Extensive benefits transfer.
Working Group on Public Health and Fossil Fuel Combustion, 1997	Global	All	OECD nations use 1990 Clean Air Act Standards; non OECD nations use same standards for mobile sources by 2020 and 1970 standards for stationary sources by 2020	Results not monetized	No	PM10 is sentinel pollutant; highly stylized analysis to develop global impacts
Ekins, 1996	Various	All	Alternative cases assume compliance with first SO <sub>2</sub> and NO <sub>x</sub> protocols	\$273	Yes	Synthesis of several other studies
Holmes, <i>et al.</i> / PREMIERE 1995	US	Electric	1990 Clean Air Act	\$3.22**	No	Motor Challenge Program; includes secondary nitrates, excludes ozone effects

(Table 1 continued over page)

(Table 1 continued)

Alfsen, <i>et al.</i> , 1995	Norway, EU	Electric	Alternative cases assume compliance with SO <sub>2</sub> and NO <sub>x</sub> protocols	24-452, 21+ ***	Yes, but not fully calculated	Human health, accidents, congestion, acidification of forests and fresh water lakes considered
EXMOD 1995, as derived in Burtraw and Toman (1997)	New York State	Electric	1990 Clean Air Act	\$23.79**	No	Reduced use of single coal plant in N.Y.; only PM, NO <sub>x</sub> and SO <sub>2</sub> (assuming emission cap in place); includes secondary particulates and ozone effects
Boyd <i>et al.</i> 1995	US	All	Economy-wide carbon tax	\$39.79	No	Human health and visibility effects calculated from reduced emissions of all criteria pollutants
Barker, 1993	UK, US, Norway	All	Various cases with alternative assumptions about meeting targets	125-282; 332, 254-386***	No	Human health, traffic accidents, congestion considered
Goulder/Scheraga and Leary, 1993	US	All	Economy-wide carbon tax with stabilization at 1990 levels in 2000	\$33.36**	No	All criteria pollutants, no secondary particulates or ozone
Dowlatabadi <i>et al.</i> /PREMIERE 1993	US	Electric	Pre 1990 Clean Air Act	\$2.95**	No	Seasonal Gas Burn; includes secondary nitrates, excludes ozone effects

\* Previous studies based on fixed emissions ratios are generally excluded from this Table.

\*\* Calculated by Burtraw *et al.* (1999).

\*\*\* As reported in RIVM (2000).



### **3. Baseline issues for assessing ancillary benefits**

We have identified five issues where baselines could be significant in assessing ancillary benefits. These issues, which are distinct from those that are generally considered in the baselines of large-scale economic models, include: policies/regulations as they specifically affect the different types of ancillary benefits, technology development and diffusion, demography other than the aggregate population, economic activity other than aggregate measures of performance, and natural baselines, including the assimilative capacity of natural systems. Other categories may also be relevant.

#### **3.1 Types of baselines**

##### **3.1.1 Policy**

Current and assumed future laws, policies, and regulations (and degree of compliance) are critical factors for assessing the relevant ancillary benefits baseline. Although environmental policy is an important element of the baseline it is not the only relevant area of concern. In fact, policies that indirectly affect baseline emissions of ancillary benefits may be as important to consider as those directly governing emissions. Depending on the issue, one can imagine that health policy (e.g., policies aimed at universal or improved quality of health care), transportation policy (e.g., CAFÉ standards), agricultural policy (e.g., nonpoint source standards which might affect water runoff and thus baseline water quality), energy (e.g., voluntary program designed to encourage the dissemination of high efficiency motors in industry such as DOE's Motor Challenge), economic regulation (e.g., deregulation of the electricity generation), and tax policy (e.g., proposed EU carbon/energy tax or US BTU tax) could all have significant impacts on the nature and magnitude of ancillary benefits.

To date most of the studies explicitly addressing policy baselines have focused on air pollution issues. However, a few studies have been conducted in other areas as well, including energy policy [e.g., Holmes, *et al.* 1995; US DOE and US EPA, cited in Dower and Morgenstern (1997)], and transportation (Barker 1993). A number of these policy areas are inter-related. For instance, to get a good fix on the air pollution benefits of GHG policies in the U.S. one needs to make assumptions about a broad array of issues affecting the pace and consequences of electricity restructuring-capturing effects on fuel mix, activity levels, abatement technologies, and the spatial distribution of these effects. If one categorizes electricity restructuring as a GHG reduction policy, then any emissions reductions (or increases) from restructuring would count as ancillary benefits (costs). At a minimum, such policy assumptions need to be explicit.

### 3.3.2 *Technology*

Assumptions about the economy-wide rates of technology/efficiency improvements are usually transparent in the macro-level analyses of the costs of GHG reductions, e.g., the AEEI (autonomous energy efficiency increases). However, these assumptions may not be sufficiently detailed to credibly estimate future baselines for ancillary emissions. Often, the effect of the economy-wide assumptions on future baseline emissions is not transparent, and sometimes it is not even addressed. For instance, assumptions about the expected rate of vehicle stock turnover, fuel quality, and the decay rate of catalytic converters as a car ages are all critical components for estimating baseline ancillary emissions, but are not generally stated or even addressed in ancillary benefit calculations. Care also must be taken to be sure that the ancillary emissions are consistent with assumptions about the future mix of gasoline, diesel, and vehicles using other types of fuels. Also, there is evidence that emissions are lower in vehicles with higher gas mileage (Harrington, 1997).

### 3.3.3 *Demographic*

While the large-scale economic models routinely consider overall population trends they generally do not take account of a number of other demographic factors that are important to the consideration of ancillary benefits. For example, continued improvements in the health status of the population will effect the estimation of ancillary benefits in a number of ways. Healthier people are less susceptible to many environmental conditions, including some types of air and water pollution. The willingness to pay (WTP) for health improvements may also vary with the health status of the population, although a recent literature review suggests that such differences only become significant above risk levels of 50 per cent (UK Department of Health 1999). At the same time, increasing urbanization tends to expand the size of the population exposed to high pollution levels while the growing elderly population tends to increase susceptibility to air pollution damages.

Although these elements are sometimes considered in economic benefit studies, to our knowledge none of them has been explicitly incorporated into the ancillary benefits literature. The sole exception is Burtraw *et al.* (1999) which included population projections according to geography, age and income in their analysis.

### 3.3.4 *Economic*

Assumptions about baseline levels and growth rates of aggregate economic activity (GDP) are essential to understanding the direct benefits and costs of GHG mitigation policies. However they may not be sufficient to understanding the ancillary benefits. Disaggregation at the industry level – already present in a number of the large scale models – is clearly critical to understand shifts from pollution-intensive industries to the service sector. In addition, to get a full understanding of the ancillary benefits it is also important to understand the size of the population exposed to conventional pollution. This, in turn, requires information on the spatial location of the emissions vis-a-vis the population. As noted earlier, the level of geographic detail varies considerably among the ancillary benefit studies. Certainly none of the large-scale CGE models are capable of addressing geographic issues.

### 3.3.5 *Natural Activities*

Our final baseline category concerns the natural baseline, particularly the assimilative capacity of the natural system. Consider the case of the Adirondack watersheds, which are particularly sensitive to potential acidification from atmospheric deposition of sulfur and nitrogen, in part because of the cool temperatures, shallow soils with low base saturation, short growing seasons, and the long history of elevated sulfur and nitrogen deposition. Because of the scientific uncertainty in the estimated time to nitrogen saturation, the range for the future baseline of chronically acidic lakes is enormous. One recent study found that if saturation was assumed at 50 years then the percentage of lakes that are chronically acidic – 19 per cent now – could more than double by 2040. Alternatively, if it is assumed that saturation is never reached, the percentage of chronically acidic lakes could fall to 11 per cent or less by 2040.<sup>32</sup>

Clearly, such wide swings in assumptions could have major implications for ancillary benefit estimates. Interestingly, recent studies in China find that nitrogen deposition into surface water transforms natural toxins into potent carcinogens. Although limited quantitative work has been done on this issue, it suggests the possibility of other types of health and ecological benefits associated with reduced nitrogen deposition (Wu *et al.*, 1998). Overall, it is clear that both basic and applied research are needed in this area.

## 4. **Treatment of baseline issues in the ancillary benefit literature**

In general, the early ancillary benefits papers explored the broad conceptual issues and developed preliminary estimates with only limited specificity about the baseline. Several recent papers have explored the baseline issue in greater detail, although the applications have generally been limited to environmental health.

In considering how the baselines affect the estimates of ancillary benefits, it is important to consider several aspects of the studies. Perusal of Table 1 indicates that the geographic scope of the studies ranges from county-specific (Los Angeles) to global, although the preponderance of the papers focus on the US. As noted, a number of the studies use disaggregated sector-specific models of the electric utility sector which include significant geographic detail as opposed to the more aggregate models of the entire economy.

The studies reported in table 1 focus almost exclusively on health issues. Inasmuch as health benefits generally represent a large portion of the known benefits of pollution reduction (70-90 per cent), this is certainly a good place to start, although it is not necessarily the only important ancillary benefit. Second, environmental policy baselines are the principal if not the only type of baselines explicitly considered in this literature. None of the studies explicitly considered ancillary benefit baseline issues for nonenvironmental policies (e.g., energy, transportation, etc) or for nonpolicy areas (e.g., technology, demographic, economic or natural activities), although some of these issues are mentioned. Thus, this literature should be seen as representing an important category of ancillary benefits but not the only one.

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<sup>32</sup> The information in this paragraph on the Adirondack watershed is drawn from a background paper by Cook, *et al.* (1999).

In the US several of the recent studies explicitly considered at least one and, in some cases, multiple aspects of the 1990 Clean Air Act. The SO<sub>2</sub> program is the most commonly modeled of recent policies, although it is not consistently handled in the literature. Some of the early studies examine the possibility of further SO<sub>2</sub> reductions associated with modest GHG mitigation policies. The more recent papers recognize the fixed nature of the cap although, as noted, only a few of the papers attempt to calculate avoided costs. Only one paper goes beyond the 1990 Clean Air Act to consider recent efforts to control interstate transport of NO<sub>x</sub>. This is an important addition to the literature because NO<sub>x</sub> is a key source of secondarily formed fine particles which, in turn, have been linked via epidemiology studies to elevated mortality rates. In Europe the Second Sulfur Protocol was not been explicitly modeled in any of the papers reviewed by Ekins (1996), although the First Sulfur Protocol was considered in several studies. One of the Chilean papers considered full implementation of the Decontamination for the Metropolitan Region (Santiago). In the single global study specific assumptions were made about the technologies in place in different regions at future dates.

Not surprisingly, small differences in the policy baselines can yield large differences in the value of ancillary benefits. Although there are few examples of sensitivity analyses designed to test the importance of baselines, some evidence is available on this point. Thus, when full account is taken of the Second Sulfur Protocol, Burtraw *et al.* (1999) estimate that the mean value of the ancillary benefits calculated by Ekins (1996) for European nations declines by about one-third.

In the SO<sub>2</sub> program a facility that reduces its emissions below its own regulatory limit displaces the need for abatement at another facility (or at its own facility in the future). The CGE models do not consider this issue at all. Only the sector-specific models are capable of addressing this issue adequately. The benefits of avoided investments in SO<sub>2</sub> abatement are generally additive with respect to other categories of benefits. Burtraw *et al.* (1999) found that for moderate carbon policies that leave the SO<sub>2</sub> cap unaffected, SO<sub>2</sub> abatement cost savings of about \$3 per ton carbon reduced should be added to the \$3 per ton ancillary benefits identified with the modeled NO<sub>x</sub> reductions. In this instance consideration of the avoided abatement costs effectively doubles the estimates of the ancillary benefits for the case of moderate carbon policies.

Other than the previously cited EPA study, the only other analysis to explicitly consider these avoided costs of regulation is by Lutter and Shogren (1999). In a widely-circulated memo, Lutter and Shogren consider the avoided cost of abatement in the context of a “capped” program very different from the SO<sub>2</sub> cap, namely the criteria air pollutant program. Unlike most of the other studies listed in Table 2, they present a fairly simple analytical formulation of how changes in carbon emissions affect the emissions of conventional pollutants. Their empirical work focuses on cost savings in attaining the newly proposed ambient particle standard PM<sub>2.5</sub> (particulate matter of less than 2.5 microns in diameter) in Los Angeles. They focus on Los Angeles as the marginal area since, as they note, “[it] is likely to be the ‘last’ area to come into compliance with EPA standards.”

Overall, Lutter and Shogren estimate avoided costs of about \$300 per ton, although there are a number of reasons to question their findings. First, the PM<sub>2.5</sub> standard is not actually in effect since it has been blocked by the D.C. Circuit Court of Appeals. However, even if the PM<sub>2.5</sub> standard as originally promulgated were to become effective, full compliance would not be assured.

For the  $PM_{2.5}$  and other costly standards, care must be taken to consider the degree of likely compliance when developing baselines. The international literature is replete with anecdotes about countries with strong laws on the books but weak enforcement. Certainly one would not want the baseline assumptions for some countries to be taken from too literal a reading of environmental laws in those countries. Yet, even where there is a strong history of enforcement some of the same problems can exist. While the US acid rain program is an example of a program with a very strong performance record, the story for other programs is less sanguine. For example, more than half the US population lives in areas that are currently in violation of the ambient ozone standard. Full compliance is not anticipated for decades, at best. Thus, it would be imprudent to assume full compliance with the ambient ozone program when estimating either ancillary benefits or avoided costs. Thus, even apart from the precarious legal nature of the  $PM_{2.5}$  standard, we think it is imprudent to assume, as Lutter and Shogren do, that it would be fully implemented in Los Angeles or in other high cost areas over the next several decades.

A further area of concern about the Lutter and Shogren calculations concerns the likely carbon reductions to occur in Southern California as a result of at least modest GHG mitigation policies. Models of the US economy estimate that under an economically efficient control regime about two-thirds of reductions are likely to come from the electricity utility sector. Yet, at present there is no coal based generating capacity in Southern California.<sup>33</sup> Thus it is unlikely that GHG mitigation policies would generate the  $PM_{2.5}$  reductions in the Los Angeles area that would be consistent with the ancillary benefit estimate of \$300 per ton of carbon even if the standards were in place and enforced in Southern California.

Our conclusion on this point is the simple one that all standards, even all capped standards, are not alike in terms of their effect on ancillary benefits. Our suggestion is that analysts should consider the degree of likely compliance with individual standards, especially costly ones, when developing baselines for either ancillary benefits or avoided costs.

A final concern about baseline issues arises when a GHG mitigation policy is associated with certain negative ancillary benefits. In the developing country context the most relevant example concerns a GHG mitigation policy that has the (unintended) effect of slowing the pace of rural electrification, e.g., by raising the relative price of electricity vis-à-vis other (unregulated) energy sources. A well specified baseline would presumably capture the expected growth in rural electricity use and the corresponding substitution away from dirty and inefficiently combusted fuels, e.g., coal or biomass. Simulation of a model with such a baseline would properly capture the negative ancillary benefits associated with the GHG mitigation policy (e.g., higher morbidity and mortality associated with the increased indoor and the outdoor exposure to fine particles and other pollutants). Interesting work in this area has recently been carried out by Wang and Smith (1999).

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I am indebted to Dallas Burtraw for pointing this out to me.

## 5. A scorecard for estimating ancillary benefits

### 5.1 *Recommendations*

Recognizing the complex issues and multiple objectives involved in establishing a credible and comprehensive baseline, it is tempting to create a type of scorecard to heighten awareness to the critical problem areas. Such a scorecard could involve tabular or other visual presentation for developing both quantitative and qualitative information on key baseline issues. In this section we begin that task by distilling the key problem areas identified in the previous sections and developing a series of recommendations to guide the preparation of ancillary benefit estimates.

As a starting point, we refer to Table 2, which arrays the multiple ancillary benefit areas discussed in Section 2 along the horizontal axis and the various baseline issues presented in Section 3 along the vertical axis. The Xs refer to areas where previous studies have been identified. The obvious temptation to call for more information or analysis must be considered in a value of information framework. In the end, of course, there is no substitute for judgement on the part of the analyst concerning the relative importance of the issues identified below.

*Recommendation #1: Consider as many types of ancillary benefits as practicable*

This is probably the most basic issue of all. Referring to Table 2, we have identified four potentially important categories of ancillary benefits: health, environmental, economic/welfare, and safety. In addition, other categories may be relevant. To date most of the research has focused on health, while limitations of both methods and data have constrained the ability to estimate the other benefit categories. More research is needed in these areas.

*Recommendation #2: Consider as broad arrow a set of baseline issues as practicable*

We have identified five areas of potential concern for the development of baselines: policy/regulatory, technology, demography, and economic and natural activities. Others may be relevant as well. Each of the baseline issues, in turn, has sub-elements that may be important. When considered in light of the multiple categories of ancillary benefits, there could be as many as one hundred (or more) baseline issues of concern. Of course, not every issue applies to every GHG mitigation policy, and many of the potential issues in Table 2 have not been seriously examined in the literature to assess their quantitative importance. Nonetheless, it might be appropriate for researchers to at least consider a list of this sort. Clearly they would choose to reject many of the categories as either unknown, unknowable or irrelevant. But the very act of considering such a list might spur new research ideas.

Table 2. **Ancillary benefit/baseline issues not generally included in GHG models**  
Categories of Ancillary Benefits

(1) Baseline Issues	(2) Health	(3) Ecological	(4) Economic/ Welfare	(5) Safety related	(6) Other*
Policy/Regulatory					
Environmental					
Air	X	X	X	X	
Water					
Waste					
Other					
Health					
Transportation	X			X	
Agriculture					
Energy					
Economic Regulation	X				
Tax					
Other					
Technology					
Innovation					
Diffusion					
Demography					
Population Health Status					
Spatial Distribution					
Age Distribution					
Economic Activity					
Subsectoral Composition					
Spatial Distribution					
Natural Activity					
Other					

\* Categories suggested in the literature include employment, technological change (which could lead to an increased GDP growth rate), community severance (e.g., loss of neighborhood due to heavy traffic flows), and others.

Recommendations #1 and #2 apply to all ancillary benefit studies of GHG mitigation policies. They represent a call for care and completeness when thinking through potential ancillary benefit issues. Naturally, these recommendations must be tempered by the costs and the benefits of developing estimates according to multiple ancillary benefit/baseline categories. In the one ancillary benefit area which is most developed in the literature - namely health - a more specialized set of issues apply.

*Recommendation #3: When focusing on air pollution issues consider the full set of relevant pollutants and source categories*

Often only a subset of the relevant pollutants are considered in ancillary pollutant studies. The early ancillary benefit studies tended to focus on a limited number of pollutants. It is now widely recognized that multiple pollutants may yield significant ancillary benefits. The more recent US and European studies have focused on NO<sub>x</sub>, SO<sub>2</sub>, and PM<sub>10</sub>. Given the importance of NO<sub>x</sub> for the formation of fine particle (secondary pollutants), this is a critical addition.

Of course, pollutants of interest can vary significantly by country. For example, in some developing countries where direct combustion of coal is still prevalent in the household sector, both indoor and outdoor exposures may be important. Similarly, there may be significant ancillary benefits associated with reduced lead exposure in a country like Chile where lead continues to be widely used as an octane booster in gasoline.

*Recommendation #4: When new or anticipated standards are considered in the baseline, consider the precise form of the standards*

When an emission standard takes the form of a hard cap, as the case of SO<sub>2</sub> emissions covered under the Clean Air Act, it is particularly important that the models accurately reflect the precise form of the standard. As several US studies have shown, because of the nature of the cap, moderate carbon policies do not generally induce further SO<sub>2</sub> reductions. In those cases, however, it is important to calculate the avoided costs associated with the GHG mitigation policy. Of course, with more aggressive GHG mitigation policies it may become economic to reduce SO<sub>2</sub> emissions below the cap, although the precise level where the cap is breached has not been fully explored in the literature.

*Recommendation #5: Be sure to consider the anticipated degree of compliance with new (or existing) standards*

It is not generally appropriate to assume that all emitters will be in full compliance with new or existing standards. Although this is not always a distinction of importance, an inappropriate assumption about compliance can introduce significant bias into the estimation of ancillary benefits. In the U.S., for example, with particulate and ozone standards now under revision, and with new emphasis on reducing NO<sub>x</sub>, it is particularly important to include new and anticipated standards in developing baselines. Failure to include such standards will tend to overstate environmental savings. It may also overstate the costs of the CO<sub>2</sub> reductions by overstating the opportunity costs of potential substitution away from old technologies. Which one of these factors is greater depends on the relative costs, elasticities, etc. In one case reviewed here, the authors assumed that all polluted areas in the US would be in compliance with the proposed PM<sub>2.5</sub> ambient air standard in little more than a decade. Given the history of compliance with other costly ambient standards in the US, we do not believe this is a realistic assumption for purposes of calculating ancillary benefits.

*Recommendation #6: Consider the location of the expected reduction in pollution in developing estimates of ancillary benefits*

As the social costing literature has vividly demonstrated, the benefits of emission reductions can vary tremendously depending on the spatial location of emission reductions vis-a-vis the proximity of the exposed population. Meteorology and other factors affecting the transformation from emissions to pollutant concentrations can also be critical. It is thus vitally important to consider the location of contemplated emission reductions when making estimates of ancillary benefits.



*Recommendation #7: Use disaggregated, geographic-relevant models whenever possible in order to capture the complex effects underlying ancillary benefit estimates*

Because of the underlying complexities of industry and geographic-specific factors, disaggregated models represent a superior approach for developing accurate estimates of ancillary benefits. Aggregate models, which have many advantages for the study of GHG mitigation policies, are not well suited to capture the important nonlinearities involved in estimating ancillary benefits.

*Recommendation #8: Beware of the tendency to overstate baseline emissions*

There may be a (natural) tendency to overstate baseline emissions of conventional pollutants. While it is difficult to demonstrate the accuracy of emission baselines in ancillary benefit studies, two observations can be made. First, as shown in table 1, if one omits the Lutter and Shogren (1999) analysis, there is a clear tendency for the estimated ancillary benefits to decline over time, at least in the US literature. Inspection of these studies suggests that one of the principal reasons for the downtrend in estimates is the secular refinement in the baseline estimates. It is probably fair to say that from today's vantage point the early estimates of ancillary benefits implicitly overstated the growth in future emissions. The more recent estimates have reduced the estimates of baseline growth and, correspondingly, reduced the estimates of ancillary benefits.

A second piece of evidence on this point comes from a recent review of the accuracy of cost estimates used in environmental and occupational regulation, including a breakout showing the accuracy of the baselines. Harrington *et al.* (2000) assembled a dataset consisting of 28 environmental and occupational regulations or policies for which detailed *ex ante* as well as *ex post* cost estimates had been prepared. About half of the regulations were issued by the US EPA with the remainder issued by the US Occupational Safety and Health Administration (OSHA), the state of California, Canada, Norway and Singapore. Most of the cost analyses applied to relatively large rules and reflected considerable effort on the part of the regulatory agencies to prepare the estimates.

The results suggest a tendency – albeit not an iron clad rule - for the costs of regulations to be overestimated. One of the reasons for this overestimation is that the quantity of emissions reductions associated with the regulations tends to be overestimated as well. Inaccurate prediction of emissions reductions can occur through mis-estimation of the baseline emissions that would exist without the regulation or through compliance problems. Cited examples of baseline errors in the Harrington *et al.* study include the US SO<sub>2</sub> program. In that case the analyses did not foresee an estimated two million tons of reductions that occurred as a result of railroad deregulation and other factors unrelated to the EPA regulations (Carlson, *et al.*, 2000). The authors note that “...curiously, our data-set contains no baseline underestimates.” In fact, they suggest, “...agencies may have a strategic interest in enhancing the potential seriousness of the problems they are regulating.”

Research in this field is limited and the cited study – which contains the largest sample of any published study – should be seen as preliminary. Nonetheless, the lesson is clear: beware of overestimating baseline emissions.

*Recommendation #9: Consider as many uncertainties as practicable when developing ancillary benefit estimates*

This is an obvious but critically important recommendation. With or without credible information on the issues described above it is essential that explicit uncertainty analyses be conducted. Even the simple use of sensitivity analysis can be helpful in highlighting the key uncertainties.

## **5.2 *Integration between economic modelers and ancillary benefits experts***

There has generally been a lack of interface between large scale economic modelers and ancillary benefits experts. More than any other single event, the third assessment report (TAR) of the IPCC has served to raise awareness in the modeling community about the importance of these issues. Nonetheless, the current state of intellectual exchange between the large scale economic modelers and the ancillary benefits experts is quite limited.

Recently, the IPCC Special Report on Emission Scenarios (SRESS) (IPCC, 1998) has been the subject of extensive discussion and review concerning the characterization of future GHG emissions. Little attention has been focused on the emission scenarios for nonGHGs or on the relevance of these scenarios for the calculation of ancillary benefits. Overall, there are a number of helpful elements in these scenarios, although many questions arise concerning their usefulness to the modeling activities currently underway in the IPCC.

The helpful elements in the scenarios concern the number of nonGHGs considered and the spatial detail available for these scenarios. SRES developed emission scenarios for four nonGHGs: SO<sub>2</sub>, NO<sub>x</sub>, NMVOCs, and CO. Annual emission estimates are available for four global regions *and* for one degree by one degree grids within those regions, 1990-2100. These estimates, in turn, are derived from the group of models used by SRES: AIM (National Institute of Environmental Studies, Japan); ASF (ICF Kaiser, USA); IMAGE (RIVM, the Netherlands); MESSAGE (IIASA, Austria); MARIA (Science University of Tokyo, Japan); and MINICAM (PNNL, USA). Not surprisingly, SO<sub>2</sub> has been the most carefully modeled, although all four gases have been studied in some detail. Limited information has been published on the construction of these scenarios but, at least in the case of the SO<sub>2</sub> emissions, they are tied to an income-based parameterization – a so-called Kuznets curve (Smith *et al.*, 1999). Under these scenarios emissions follow a ‘U’ shaped pattern wherein emissions first increase with increasing use of fossil fuels (mostly coal) and then decrease as controls are implemented. In the IPCC parameterization controls are adopted somewhat more rapidly than would be predicted on the basis of the historical growth in GDP per capita in the US and other developed countries. The one degree by one degree grids are based on an apportionment of emissions within the four regions modeled.

Unfortunately, at least on the basis of the published information, there is a lack of transparency about a number of key elements of these scenarios. While a Kuznets curve approach is appropriate for modeling long term trends (e.g., one hundred years), it is not particularly use for understanding the specific policies captured in the baseline. For the purposes of estimating ancillary benefits, it would be helpful to develop more detailed baselines for the nonGHGs for a shorter period of time (e.g., twenty years).

Finally, there is the critical question of how to integrate the information on nonGHG emissions into an economic analysis of ancillary benefits of GHG mitigation policies. The clear advantage of the large-scale economic models used by the IPCC is their ability to incorporate general equilibrium effects not available in the simpler models. However, given the level of aggregation of these models, it is not currently possible to develop ancillary benefit estimates which incorporate the industry and spatial detail needed to assure accuracy of these estimates. One recent CGE modeling effort in Sweden considered ancillary benefits, although it only developed estimates of emissions of nonGHGs as opposed to actual benefits (Nilsson and Huhtala, 2000). Yet it is the calculation of benefits that involves the more complex analytical and data issues

The SRES scenarios represent a positive step forward in their handling of nonGHGs. However, there is along way to go before even that level of disaggregation can be used to calculate ancillary benefits in the large-scale economic models. For the purposes of policy analysis, it remains an open question as to how best to integrate the two modeling approaches, at least in the near term.

## **6. Conclusions**

Several points emerge from this review of baseline issues relevant to the estimation of ancillary benefits.

Ancillary benefits of GHG mitigation policies could potentially offset a significant portion of the costs of those policies. Inadequate consideration of these benefits could lead to the selection of GHG mitigation policies of inappropriate stringency or design. However, developing sound estimates involves many complex issues.

Baselines are an important component of ancillary benefit estimates. Small changes in baseline assumptions can lead to large changes in estimates of ancillary benefits. Currently, baseline issues are not addressed in a consistent manner in ancillary benefits studies. Problems of double counting as well as undercounting can be found in the available literature.

1. Most of the focus in the literature has been on ancillary health benefits associated with the reduction of conventional pollutants. Other categories of benefits e.g., ecological, economic/welfare, and safety need to be considered. Even when monetary valuation of these benefit categories is not possible, effort should be made to treat the outcomes as explicitly as possible, including the provision of both qualitative and quantitative information.
2. Only one element of the five potentially important baseline issues, namely, government policies and regulations, has been systematically treated in the ancillary benefits literature. Even there, however, the focus has been on environmental policies. Other policy baseline issues (e.g., energy, transportation, health, etc) have been generally ignored, as have the nonpolicy baseline issues (e.g., technology, demography, natural). All these need to be considered in an explicit manner in order to develop more credible ancillary benefit estimates.
3. Principally because of the importance of the location of emission reductions and of exposed populations, highly disaggregated models are the preferred tools of analysis for estimating ancillary benefits. Large errors can be introduced into the calculation of ancillary benefits by failing to consider issues of spatial location of emissions vis-à-vis potentially exposed populations.

4. As the ancillary benefit literature has evolved over the past decade estimates of ancillary benefits have declined somewhat. This is due in large part to the consideration of better articulated baselines and the use of the more disaggregated models which are able to incorporate the better articulated baselines. For example, current policies in the US are targeting individual power plants for NO<sub>x</sub> reductions according to their contribution to ambient pollutant concentrations and the cost of making reductions at those power plants. To accurately represent such policies in the baseline – and thus avoid double counting - requires a highly disaggregated model with considerable spatial detail.
5. There are large information and modeling gaps, particularly concerning the spatial location of ancillary emissions. These gaps are especially large in developing countries, where it is difficult to model secular changes in the baseline in a sufficiently fine-grained manner, to link the results to current policy developments. The use of Kuznets curves for modeling such emission scenarios is a promising approach, although the connection to current and planned policy developments is often tenuous.
6. The need for highly disaggregated and spatially relevant models to adequately incorporate the relevant baselines may conflict with other goals for the analysis of GHG mitigation strategies. For example, large CGE models which are used for cost estimation operate at a completely different spatial scale than the more localized models relevant to estimating ancillary benefits. The CGE models do not and, realistically, cannot soon incorporate this type of detail necessary for making sophisticated estimates of ancillary benefits.
7. Given our current inability to integrate the spatially relevant models for estimating ancillary benefits and the large scale economic models, probably the most pragmatic approach is to continue to rely on (an expanded set of) case studies of ancillary benefits estimates. The alternative, i.e., using the aggregate economic models to directly estimate ancillary benefits, would be costly as measured in a large loss of accuracy in the ancillary benefit estimates.

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# EXTENDING INTEGRATED CLIMATE ASSESSMENT MODELS TO INCLUDE ANCILLARY BENEFITS: PROBLEMS AND PROSPECTS

by Hugh PITCHER<sup>34</sup>

## Abstract

*This paper describes the ancillary benefits problem in its broad context, and discusses critical elements that need to be added to conventional climate modeling tools to adequately describe three major areas, local air pollution, land use and water quality and supply, where climate policy may either affect or be affected by other environmental concerns. It also reviews some of the methodological issues raised by addressing these three areas, and suggests some strategies for addressing these problems. The paper concludes with some cautionary notes about the difficulties involved in integrating ancillary benefits into quantitative global climate impact models and the importance of providing for the consideration of factors which cannot currently be quantified.*

## 1. Introduction

The climate change problem arises because human-activities produce a wide variety of emissions of greenhouse gases (GHG) that are anticipated to change atmospheric behavior in ways that will increase temperature and affect precipitation and climate variability. While the bulk of the relevant emissions come from energy use, agriculture and industrial activity also emit significant emissions. Many of the activities leading to greenhouse gas emissions also emit other environmental pollutants of concern. Thus we can expect that policies to control greenhouse gases may reduce (or expand) a variety of other impacts of societal concern. It is these other changes that constitute ancillary benefits (or dis-benefits). Therefore, as we consider the development of sets of strategies to manage the climate issue, it is important to have a full understanding of the results of implementing such policies. In this regard, the tools that are used to assess quantitatively the policy alternatives for reducing GHGs need to include the ability to simulate the full range of effects, not just those that are related to climate change.

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This paper explores the development of interest in ancillary benefits by the general climate modelling community and others, discusses 3 types of ancillary benefits that can be incorporated into these quantitative models, and notes some methodological difficulties and challenges of providing for integration of ancillary benefits into the policy discussion. This is followed by a discussion of some of the kinds of strategies that could be expected to be useful in resolving these issues. The paper concludes with a discussion of the ways these kinds of tools can be used, of the resources required to develop them, and a suggested path forward.

**The full range of effects of climate policies is potentially quite broad**, since the two major sources of greenhouse gases, energy production and agriculture/land use, affect almost all aspects of human activity. Included in the policies under consideration to manage the climate problem are such diverse strategies as increasing energy efficiency, decreasing energy demand, increasing the efficiency of non-carbon sources of primary energy such as solar, providing alternative motive power for vehicles, changing agricultural management practices to increase storage of carbon in the land, as well as managing the way in which diets change as incomes increase in the developing countries.

The dominant focus for work on ancillary benefits has been local air pollution. This focus arises because energy production and use is a major source for both problems. But the two problems are not identical, having very different patterns of emissions and policy impact profiles. In addition to local air pollution, there are two other major areas it is useful to consider. The first, land use, arises because some policies to reduce or offset carbon emissions assume large scale changes in land use, for example, biomass, or in land management to increase carbon in land. The second is water, whose supply is likely to be changed significantly by climate change, and whose availability and quality may also be compromised by various proposed energy policies. These three areas do not exhaust the topics of interest, but they will illustrate the range of geographic scale issues and time step issues that need to be resolved to construct integrated programs to manage a wide variety of environmental issues.

**In addition to climate policy impacts on other environmental issues, it is essential to consider the impact of other policy on climate results.** For example, reducing fine particulates by limiting sulfur emissions from electricity generation is anticipated to increase temperature change, although the magnitude of this effect remains uncertain. Limiting emissions of local air pollutants by introducing a new generation of internal combustion engines could foreclose an alternative source of motive power, fuel cells, considered to be an important tool to reduce carbon emissions.

The analytic systems used for examining ancillary benefits have to be useful in a variety of communities. Initially, the issues have to be examined and tested within the science and research communities. After these communities have considered the tools and results, the policy community uses the tools to determine what alternatives might be appropriate. Finally, the political process, which necessarily includes the public at large, has to consider and decide on appropriate responses. In the light of the diverse groups that can be expected to use results from the modeling exercises, the modeling tools must not only be scientifically defensible, they also must be transparent, flexible, and reliable.

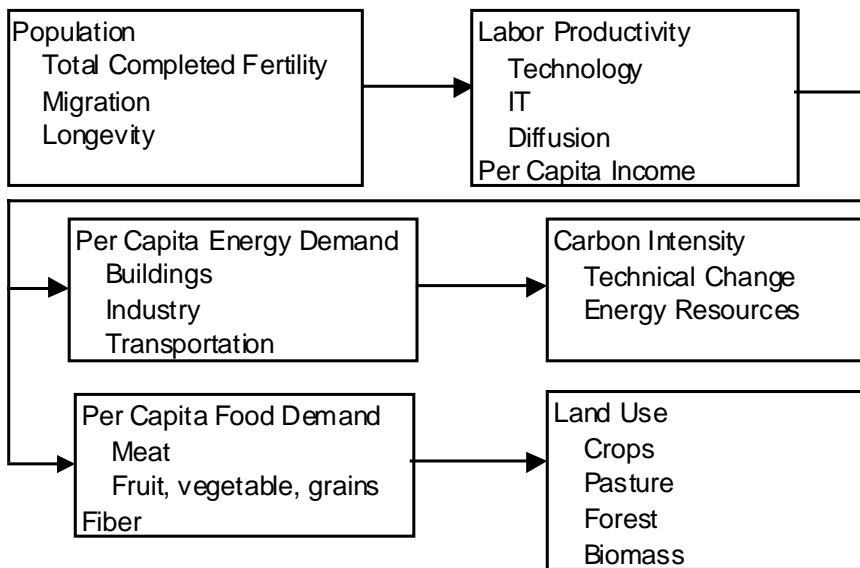
**The existing tools for analyzing the climate problem** have begun to explore the wider set of issues that are raised by the ancillary benefits problem, making the theme of this paper, how to model such changes, clearly appropriate and important. Major systems presently under development include IMAGE(Alcamo, et al, 1998), AIM(Jiang et al, 2000), EXTERNEE (Barker, 2000) and PCAM(Edmonds et al, 1993).

**The considerations discussed above imply that introducing ancillary benefits into the decision making process for climate change significantly increases the complexity of the policy environment.** One cannot simply add the benefits of the non-climate related changes due to controlling greenhouse gases to the benefits of the climate changes and compare these to the costs of the climate policies. Because resources to manage the environment are limited, introducing ancillary benefits requires adding consideration of alternative policies to achieve the ancillary changes, such as alternative ways to control local air pollution. The policy problem is then one of allocating the available resources so as to most effectively solve a wide variety of environmental problems, which in addition to the climate problem will commonly include public health, air and water pollution, as well as solid waste and land use management. While it is clearly appropriate to consider this wider set of problems, effectively doing this remains one of the major challenges to environmental management.

### **Three kinds of ancillary benefits: air pollution, land use changes, and water quality and supply**

The introductory section suggested that there were three major areas in which ancillary benefits could be expected to be important. In addition, it was suggested that these three areas would introduce systematic problems with geographic scale and time steps. To understand why this is the case we begin by introducing the following simple framework for the sources of greenhouse gases. As shown in Figure 1, the first important determinant is the scale of economic/human activity which is driven by population and worker productivity. Figure 1 suppresses important issues with respect to the geographic allocation of this activity and the inequality of income. Figure 1 also shows the two important processes which generate economic activity, the demand for and production of energy, and the demand for and production of food and fiber. The green house gases produced by these two systems are well mixed in the atmosphere, and therefore it is not important where they are produced. Thus the energy and land uses sources for these gases need only be modeled at a convenient scale in terms of the economic activity involved, which typically represents the entire world with a breakdown of from ten to twenty regions. In addition, very aggregate time steps, which range from five to fifteen years, are typically used, because these are sufficient to understand the long term trends in population, economic activity, and technological characteristics which is all that is necessary to understand how greenhouse gases are changing. These long time scales are also sufficient to understand the long terms changes in global temperature change, and in sea level, the two indicators most often used to measure climate change.

Figure 1. **Simple framework for sources of GHG**

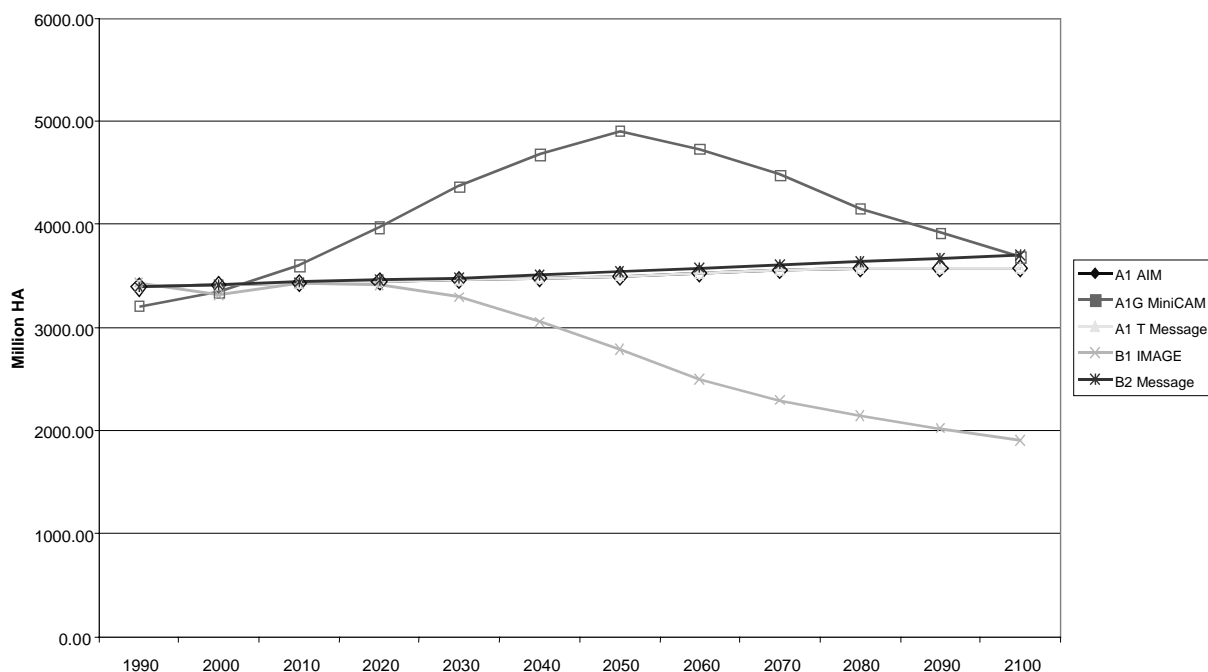


Understanding local air pollution, land use and water issues requires very different geographic breakdowns and time scales from those used in typical climate tools. Local air pollution is driven by emissions on the one hand, and by the pattern of local weather on the other. Understanding this issue requires both an understanding of where economic activity is occurring and how the weather is changing on a much a much finer scale than is needed for modeling climate effects. Not only that, most of the severe air pollution problems are of a very short term duration and a function of particularly unfavorable stable air masses, implying we need a rather fine time step for our models, and the need to have climate models provide information about changes weather patterns, a capability well beyond what any of the current climate models can provide. As referred to earlier, on the societal front, it is almost certain that many sources of local air pollution will be controlled as developing countries become richer, leading to reductions in sulfur emissions and other fine particulate sources, which in turn will cause significant increases in observed temperature change.

Analyzing local air pollution risks involves understanding the sources of and extent of pollutant emissions, the process by which these are transformed into concentrations, the patterns of behavior that lead to human exposure, the set of factors such as prior disease and genetic susceptibility that lead from exposure to health risk, and the societal benefits that accrue from a reduction in risk. The health risks associated with exposure are complex, since there risks are interactive in nature. Thus, reducing the health risk may be more effectively achieved by improving nutrition status than by reducing local air pollution. Different kinds of health risk respond very differently to reductions in pollutants, with acute respiratory infections responding quite rapidly, while chronic obstructive pulmonary disease responds only very slowly.

**Land use raises a different set of issues.** First, as the recent IPCC exercise to develop a new set of emissions scenarios discovered, there is wide variation in the potential future development of dietary preferences as incomes rise in the developing countries. As shown in Figure 2, there one scenario in which demand for pasture land grows rapidly, while all the others remain flat. This difference can be traced back to different projections for the demand for beef and other forms of ruminant livestock, which are typically range fed for a large portion of their life. There is already great concern about the destruction of forest and the loss of land due to desertification, erosion, waterlogging, and accumulation of salts due to improper irrigation. There is also much interest in land based activities that might play a major role in reducing carbon emissions through the production of biomass, or sequestration of carbon in soils through changes in management practices. What seems clear is that we do not yet have a sufficient understanding of the demands for land that are likely to emerge in the future to be able to assess what the potential negative results of large scale biomass might be. We also cannot assess whether it is feasible to reforest large areas as permanent stores. Nor, can we determine whether we can expect to keep large amounts of carbon in the ground through permanent changes to such management practices as no-till agriculture.

Figure 2. Pasture land in the SRES marker scenarios



**The second major issue about land is that there is some evidence that there is a major flow of carbon into the land.** However, we do not yet know in what geographic area this land based sequestration is occurring, nor do we understand the biological processes which are responsible for it. Thus, to understand the climatic impacts on land use issues, we need a much better understanding of the basic drivers for land use, the potential for improvements in management practices which are destroying the productivity of significant amounts of land, as well as a deeper understanding of the land based carbon cycle.

Finally, water poses a still different set of issues. For a large part of the earth's surface water is the factor which limits the productivity of the land. The productivity benefits of water have been understood for thousands of years and are the basis for the very extensive set of irrigations facilities found in all areas of the world. The geographic basis for understanding water issues is the river basin, still another geographic scale which needs to be considered in the models. There is good reason to believe that climate change will affect stream flow in two important ways. First, in those regions where snow pack is an important source of water storage, there may well be less snow and more rain, resulting in a smaller snowpack, which is expected to melt earlier, thus changing the annual runoff patterns. In order to maintain the same annual yield, or sustainable flow, this will require more storage, and more storage is increasingly difficult to implement, because of human, ecosystem and recreational changes created by large dams. The second problem is that while climate change is expected to increase the speed of the hydrological cycle, the increased precipitation this implies may be more than counteracted by the increased rate at which evaporation and transpiration will occur in a warmer climate. The result may well be decreased stream flow. Finally, there is a concern that the location of precipitation may change, with some models showing a decline in rainfall in many of the central continental areas, such as the Great Plains in the United States.

## **2. Methodological Issues**

The challenge posed in the previous section, to develop a set of tools which let us understand the much wider set of issues posed as we extend the climate models to consider a fuller set of environmental objectives and policy tools, is truly daunting. Typical climate based tools, commonly referred to as integrated assessment models, are already large and complex tools. As discussed, adding a capacity to manipulate air, land and water issues requires a major increase in the complexity of these programs. This poses major problems for the development, reliability, execution time, and transparency of the resulting expanded tools. In this section we discuss each of the major issues that experience suggests will arise as the tools are developed.

**Complexity:** Complexity is a function of the number of elements contained in the modeling system, and it typically will rise in an exponential fashion. In order to maintain transparency and efficiency, it becomes crucial to be very disciplined in reducing the complexity of each of the elements of the problem. Otherwise, the system becomes so complex and difficult to manage that effective policy analysis and response is not possible. An example of the issues which arise when the full complexity of the issues are considered is the Intergovernmental Panel on Climate Change (IPCC) paradigm for analyzing climate issues. This paradigm, which includes the following steps: develop emissions scenarios, analyze these emissions with global scale atmospheric chemistry and coupled atmospheric/ocean models, and consider the impacts of the resulting climate changes with a variety of tools, takes on the order to three to five years to complete with costs on the order of five to ten million dollars per scenario. Even as a research tool, the process is cumbersome and expensive. Because models of climate impacts are being devised as tools for the policy process, the pace and cost of scenario development heretofore is unworkable.

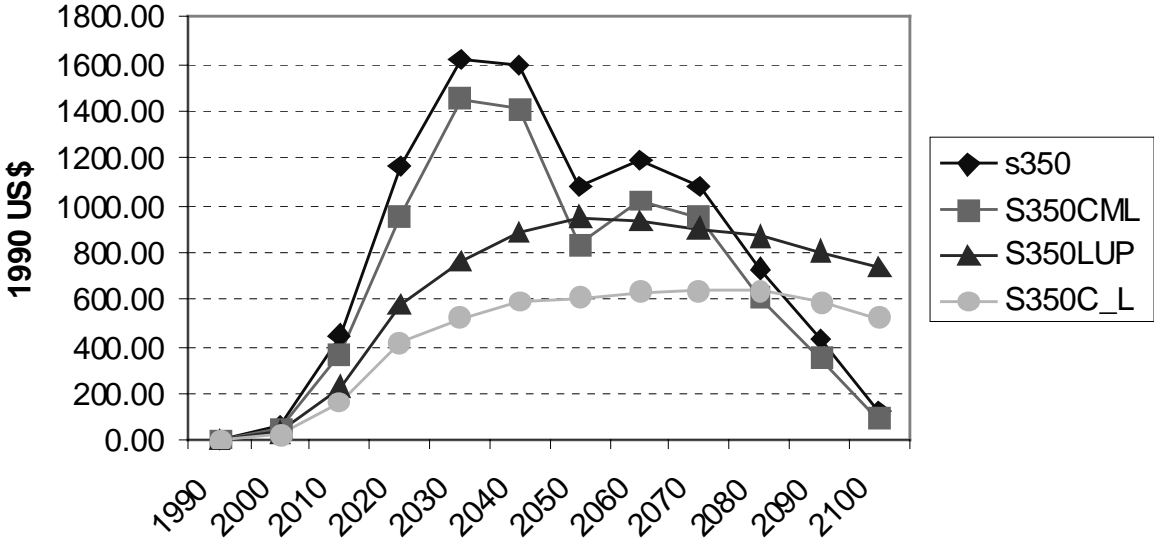
In order to provide useful information for those charged with making decisions in the near term about GHG policies, it is important to reduce the scale of the components, and to simplify the information moved between the elements of the model. There is no right answer to how to do this: the tradeoff is one of execution time and transparency against the potential cost of missing important interactions. In addition, because most decisions are made at the local or national level, it is important to generate information that has direct relevance to regional or local decision makers.

**Interactions:** The essence of modeling ancillary benefits is joining models with substantially different subjects and normally different styles, scales, and time steps. Beyond the human and practical issues which arise in these kinds of collaborative effort, the act of specifying the interfaces between the models often generates new questions, and the pursuit of these questions often leads to new insights about how the system behaves. These in turn can lead to modifications of either or both of the systems, so that the process of integrating the models leads to new scientific or policy understanding.

In a practical sense, in order to understand system behavior it is necessary to be very clear about what is happening in the interface between the models. This will often necessitate aggregating or disaggregating across physical scales, as when adding land use to an economic model, or across time scales, when integrating climate and economic models. Experience suggests that the values of variables as they pass through the interface should be readily accessible to the modeler—otherwise the process of understanding why the models behave as they do is nearly impossible.

An example from the work done for the Working Group III chapter on mitigation illustrates the kind of issues that might be expected to arise (Pitcher, 2000). As a device to test model behavior, a scenario was run whose goal was to stabilize CO<sub>2</sub> concentrations at 350 ppm. Then a subsequent scenarios was examined in which carbon sequestration from energy production was added. As seen in Figure 3, this made only a very small difference in the very high carbon tax necessary to achieve the 350 target. Investigation showed that even though carbon was being taxed very aggressively, land was being rapidly converted to pasture in order to supply meat as diets changed in response to rapid income growth. The carbon emissions from the land conversion used up more than three quarters of the allowable carbon emissions budget, leaving no space for even the small emissions remaining after sequestration. Assuming that it would be possible to reduce the large scale land conversion resulted in the carbon tax trajectory labeled land use in Figure 3, while the combination of a carbon sequestration and a land use policy (the CLU trajectory) resulted in a two thirds reduction in the carbon taxes needed to achieve the target. The important lesson this test revealed was that it was important to modify the agriculture and land use model so that behavior in this sector reflected the overall policy towards carbon.

Figure 3. **Impact of sequestration and land use on carbon taxes necessary to reach 350 ppmv**



**Stochastic Processes:** Important elements of the impacts of climate change may arise from events where current theory and practice suggests there is no reasonable likelihood of being able to improve model performance to the point of making long term forecasts. For example, current weather forecasting technology has no ability to improve upon seasonal means more than 15 days into the future. Even the variation in weather patterns produced by the climate models is often only the current long term pattern adjusted by the estimated change in temperature and average rainfall. Thus it is unlikely that weather related causes for high air pollution events can be reliably estimated, except perhaps for the impact of changes in ambient temperature. Likewise, the observation that for much of the United States there has been an increase in large rainfall events is going to be difficult to substantiate within the current framework. The same lack of precision must be attached to the likelihood of major droughts. It looks likely that projecting certain major impacts will be beyond the capability of our tools for the foreseeable future.

**Multiple Baselines:** As indicated in several parts of the discussion so far, the baseline evolution of economic and human systems is a critical component of understanding how climate policy may affect and be affected by other important environmental objectives. As the recent SRES made clear, there is no reliable way to develop a best long-term projection of the future. The unknowns are simply too large, and the evolution of critical systems too sensitive to small changes in such things as the rate of technical change in different sources for primary energy to allow such a forecast to ever be achieved. The SRES recommendation is that all analysis should use multiple baselines, as a way of being sure that the uncertainties about the future are explicitly incorporated into the analysis. Since relatively few studies and tools have routinely used multiple baselines, the methodologies for doing this and presenting the results in a coherent transparent way have not yet been made explicit or solved. In contrast, for some of the impacts of major interest for ancillary benefits, such as public health and land use, shorter time frames can be assessed with some precision.



***Some Tricks to Reduce Dimensionality:*** So far the discussion has focused on the things that need to be added, and some of the areas where it is going to be difficult or impossible to produce reliable results. But there is some hope that the system can be made to work. For many questions, making the results and the reasons for the results clear will require analysis for only a small subset of the total set and for more limited periods of time. The analysis of the impacts of climate policy is apt to be well understood with an analysis of ten exemplary cities, rather than all of the world's major cities. Once the tools are demonstrated in this way, then particular problems can be addressed with relative ease, should additional analyses be deemed important. A second approach utilizes a sample from the set of available scenarios, advocating a strategy which is used today by the large scale climate modelers of analyzing only a few of the many scenarios, and then capturing this information in much smaller models such as MAGICC, so that the essential results of the large process oriented models are available in a computationally very cheap form.

## ***2.1 Discussion***

Including ancillary benefits in integrated assessment tools results is problematic. Managing these problems is essential to obtain useful, understandable results. Perhaps the most difficult problem arises from what is known as the 'curse of dimensionality'. Incorporating more detail and more subject matter areas results in tools which are difficult to operate, difficult to debug, difficult to run and difficult to explain. It is essential to use judgement at many points in the process of developing the extended models to require that the additional material be germane and contribute to the overall system behavior in some qualitatively important way. If there are six potential arguments, but only the first three matter much to the behavior of the component, use only the first three. The constant temptation is to add more detail, the price of this temptation is the creation of tools which cannot operate within reasonable time frames and lose the important aspects of their behavior in a welter of detail. Another advantage of being parsimonious is that the results become easier to understand and interpret. It is then more likely that major elements which are not included in the analysis can be perceived and corrected for, even if by a process exogenous to the model. Being able to make such adjustments is crucial to the overall sense of reliability of the model.

The crux of the ancillary benefits issue is that policies to control greenhouse gas emissions have joint products; they also end up reducing emissions of other gases, or have impacts that are germane to other major areas of concern, such as land use patterns. Joint products have the unfortunate characteristic that it is no longer possible to assign unique costs to the various outputs. In this regard, the costs and benefits that may be compared are not developed with the same methodologies, rendering the comparison of limited value.

The discussion so far has suggested that there will be a lot of work required in order to enable the careful analysis of the impacts of climate policy on ancillary benefits and other crucial economic and environmental outcomes. Likewise, a careful understanding of how some of these other policies might work out is crucial to understanding the milieu within which climate policy will be carried out. What does seem evident is that by applying careful judgement and using simplified tools rather than the full scale process based models, it will be possible to improve the understanding of how ancillary benefits are affected by proposed GHG policies..

However, it is also important to understand that there are aspects of the problems raised by considering ancillary benefits that we are unlikely to be able to analyze quantitatively. These limits flow out of the many aspects of the models that reflect either very poorly understood but quantitatively important variable such as the average number of children born to mothers in each generation or a level of information which is simply not feasible to attain, such as detailed measurements of atmospheric conditions over the worlds oceans.

Because of the complexity of the tools involved in climate modeling, it is essential to use the results with care. Admittedly pithy phrases like ‘common sense’ and ‘grain of salt’ spring to mind. It is especially important to ask if assumptions about fixed inputs such as energy supplies are driving results and are unlikely to be fixed in the real world. To make modeling results more useful, it is essential to produce lots of graphical outputs, especially ones which show the historical and the forecast period in the same graph. Graphic displays allow inspection for the possibility of major trend breaks at the start of the forecast period, one of the sure clues that something may be amiss in the forecast period. Failure to perform these kinds of analyses can lead to major mistakes.

A particularly troublesome element in developing and understanding climate impact models is the presence of many critical areas which simply cannot be well quantified. One type of problem is that we don’t understand long term demands very well, whether it is for babies, for energy or for food. These are critical for understanding the long term levels of the economic activity. The best way to handle the uncertainty is to use multiple scenarios which reflect the plausible range of outcomes. Plausibility is a tricky concept in the phrase. As Granger Morgan and colleagues have shown, experts normally overestimate the reliability with which they can estimate effects. Because of this expert ‘hubris’, it is essential to push the boundaries a bit, especially if there is any suspicion that there may be non-linearities in the system. Another aspect of poor ability to quantify modeling results is the presence of many elements in the system which are noisy or chaotic in nature. Weather patterns, for example, are critical to predicting local air pollution, and changes in the patterns cannot be projected with any skill. Again, the solution is to use scenarios which cover a range of outcomes.

**Resources:** The additions required to effectively model and understand the issues raised by ancillary benefits are considerable. The existing integrated assessment tools are the product of extensive efforts by teams of researchers over what is approaching a decade. The process of model development, calibration, validation and data development is resource intensive. It requires a multi-disciplinary approach with considerable collaboration between those who develop specific models for such areas as local air pollution and the members of the integrated assessment group. Because of the distinct possibility that there will be unforeseen and important feedbacks, it is essential that the tools actually be integrated.

Use of the best possible tools to manage model development, input and output will reduce the difficulty of the task at hand, so that the tools are easier to develop, debug, evaluate, and manage. This will increase the reliability and value of the tools.

**Priorities:** There are many aspects of the climate problem, especially those having to do with understanding impacts, which in turn will help us to understand what constitutes ‘dangerous anthropogenic interference with the climate system’, the FCC touchstone for determining the level of effort required to reduce climate emissions, which are simply not well understood today. The issue has to be raised about the level of effort which should go into the study of ancillary benefits when the direct impacts of climate are not yet well enough understood to allow us to determine how urgent the task of reducing greenhouse gas emissions is. Some have suggested that the potential ancillary benefits of proposed GHG mitigation policies may well be greater than the direct benefits of these same policies, in regions such as Santiago (Cifuentes *et al.*, in press, JAPCA) It would seem preferable to focus on such areas as public health, food production, sea level rise, and the potential to adapt before spending large amounts of resources on the issue of ancillary benefits.

However, because control of local air pollution and the base case evolution of land use will be such important determinants of the environment within which climate policy must be conducted, a balanced approach is required that highlights clear and evident local and regional impacts.

It is essential to begin work on modeling ancillary benefits with simplified regional models, so that the major elements of the problem can be understood, and areas where further elaboration is required can be identified. As tools are generated for use in these models, it is essential that experts in the relevant science be involved who can identify those elements in the system which contribute least to the performance and can therefore be eliminated.

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# THE ANCILLARY HEALTH BENEFITS AND COSTS OF GHG MITIGATION: SCOPE, SCALE, AND CREDIBILITY

by Devra DAVIS, Alan KRUPNICK and George THURSTON

*Section 899 of the Code of Criminal Procedure for the State of New York, opens with the statement that “persons pretending to foresee the future shall be deemed disorderly...”*

## 1. Introduction

Climate change mitigation policies will bring with them ancillary or secondary benefits and costs in addition to those directly associated with avoided temperature changes. The ancillary public health effects of air pollutant emission changes (such as in reductions in particulate matter (PM)) have been the focus of some research attention. However, none of the major economic models of climate impacts has incorporated these ancillary impacts into their analyses. The extent and scale of ancillary impacts will vary with the particular Greenhouse Gas (GHG) mitigation policy chosen. The consideration of ancillary benefits can be of critical importance for devising optimal mitigation strategies. To assist in the assessment of ancillary benefits, this paper examines literature relevant to the assessment of ancillary benefits for public health of GHG policies and offers criteria for evaluating this work.

A broad array of tools is required for the conduct of public health evaluations of ancillary costs and benefits, and such tools are readily available. Energy scenarios can be employed to produce scenario based risk assessments, which rely on emission inventories and air pollution and dispersion models. Estimated changes in concentrations and exposures from these scenarios can then be linked in order to estimate incremental changes in public health that could result from various GHG policies (Burtraw *et al.*, 1996, Cropper and Oates, 1992, Holmes *et al.*, 1995, Jorgenson *et al.*, 1995, Viscusi *et al.*, 1994, Burtraw and Toman, 1997, and Burtraw *et al.*, 1999). In addition, a global assessment has been made of the impacts on public health that could arise between 2000 and 2020 under current policies, and under the scenario proposed by the European Union (EU) in 1995. (Working Group on Fossil Fuels, 1997, Davis, 1997). Country-wide assessments of GHG mitigation policies on public health have also been produced for Canada (Last *et al.*, 1998) and for China (World Bank, 1997), under differing baseline assumptions.

In addition, there are numerous cost-benefit analyses that have been used to assess the public health benefits of proposed environmental regulatory strategies (e.g., U.S. EPA, 1999, Thurston, et al, 1997). Indeed, such cost-benefit analyses are required in the U.S. by Executive Order for major new regulations. The public health effects of emissions changes in GHG co-pollutants is only one class of many potential ancillary effects of GHG reduction policies. To estimate the entire array of ancillary public health cost and benefits of alternative GHG mitigation strategies, a wide body of knowledge and practice has been developed for potential application

At the same time, both the public health and the economics literatures are rife with ongoing controversies. These include: the concentration-response functions being used (the appropriate pollutants, the nature of and uncertainty about the functional relationships (e.g., whether thresholds exist)) and, the valuation of various health endpoints (the appropriate estimates for the value of a statistical life or statistical life year, for instance). Ideally, such cost-benefit studies should include pollutants emitted into all environmental media, including air, water, and land. For instance, water and soil pollution can arise from the deposition of combustion products from fossil fuels, such as reactive sulfur and nitrogen compounds, which interact with naturally occurring humic materials, phytochemicals and byproducts of chlorination to yield highly potent and reactive byproducts (Wu *et al.*, 1999). These byproducts can exacerbate water pollution and worsen water quality, causing adverse impacts on human health.

Nevertheless, judging by the many past cost-benefit analyses of the effects of energy use on the environment and health, the largest quantifiable category of ancillary benefits or costs is related to air pollution and health effects. Of these, the largest contributor to the quantifiable monetary valuation of health benefits involves mortality risks, while the numbers of estimated adverse effects are dominated by less severe health outcomes.

While recognizing that air pollution is but one example of the potential auxiliary changes that will accompany GHG mitigation policies, this paper attempts to clarify the weight of the evidence on the important relationship between air pollution and health on the one hand, and health and economic values on the other. Annex I presents a discussion of basic epidemiologic and economic concepts. The body of this manuscript investigates, from the perspective of the health effects literature and the valuation literature, several key aspects of this relationship:

- the *scope* of ancillary effects (Section 2) —which categories of health are likely to be affected when carbon reduction policies are put in place, including both quantified and non-quantified health effects, and which of these have been expressed in monetary terms;
- the *scale* of ancillary effects and benefits/costs (Section 3 for health effects, Section 5 for valuation):
  - – How large are these effects likely to be?
  - – What is the population at risk?
  - – How large are the monetary values of these effects?

- The *credibility* of these estimates (Section 4 for health effects; Section 6 for valuation). What are the strengths, weaknesses, limitations and uncertainties characterizing the relationships between pollution and health and for the various methods of estimating the economic valuation of these effects?

We close with recommendations to policymakers and modelers on the appropriate role of ancillary health benefits and costs in the climate change mitigation debate and supporting modeling. As part of these recommendations, we also identify a number of public health and economic related research topics that require clarification in order to promote the more effective conduct of ancillary benefits assessments with respect to GHG mitigation policies. See Annex I for a discussion of basic epidemiologic and economic concepts.

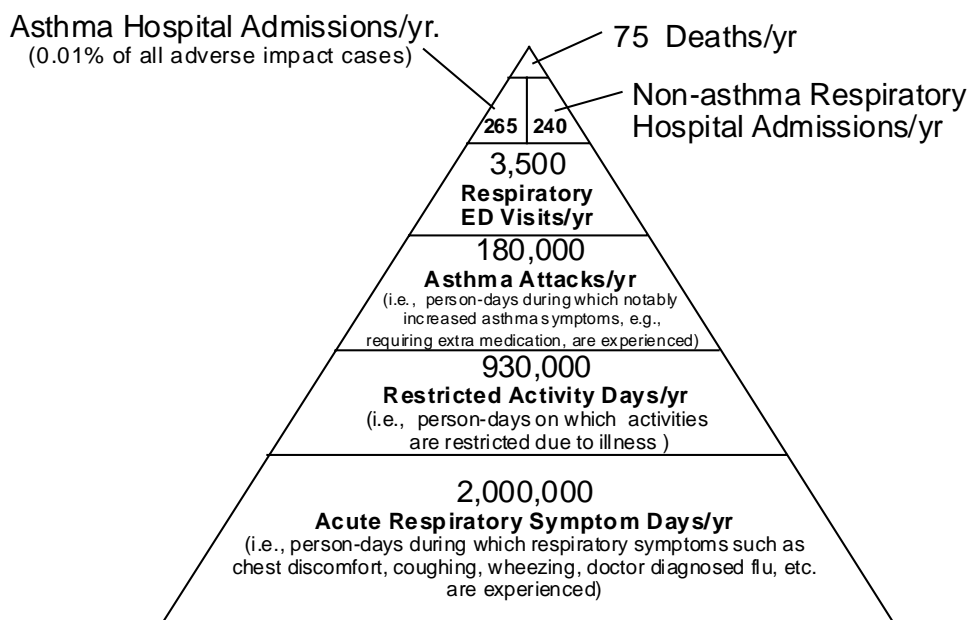
## **2. Scope of ancillary benefits**

### **2.1 Health effects**

Most of the research that has quantified health effects linked with air pollution extends across the entire human age range. Considerable numbers of human studies have been conducted for the commonplace “community” air pollutants. As a result, there is a large and rapidly growing number of studies linking air pollution with both mortality and morbidity throughout the world. A broad consensus has emerged over the past 5 decades showing that ambient levels of pollution in developed countries today are linked with measurable impacts on public health. Public health disasters, such as the episodes that occurred in London, 1952 and Donora, 1948, provided unambiguous proof of the damaging potential of high levels of air pollution for public health.

More recently, sophisticated time series analyses of daily pollution patterns, along with cross-sectional and cohort studies of differences in rates of chronic and acute illness, have provided a consistent picture of the extent to which ambient levels of air pollution affect severe health outcomes, including mortality and hospital admissions. But, as displayed in Figure 1 for the city of New York, these routinely recorded severe health outcomes represent only the “tip of the iceberg” of adverse effects associated with this pollutant. They are best viewed as indicators of the much broader spectrum of adverse health effects, such as increased restricted activity days and doctors visits, being experienced by the public as a result of air pollution exposures. Although these outcomes have much lower “monetary valuations” than the more severe impacts, their presence is important to recognize when gauging the scope of air pollution’s impact on society.

Figure 1. **Pyramid of New York City, NY Annual Adverse Ozone Impacts avoided by the implementation of the proposed new standard (vs. “AsIs”)\***



\*Figure section sizes not drawn to scale.

Source: Ozone and Particulate Matter Standards: Hearings on the Clean Air Act Before the Subcommittee on Clean Air, Wetlands, Private Property and Nuclear Safety and the Comm. on Env't and Public Works, 105<sup>th</sup> Congress (1997).

A summary of the presently quantifiable and the suspected effects of ambient air pollution on human health are summarized in Table 1. From this table, it can be seen that any air pollution impact assessment, especially in terms of numbers of health effects, will not be able to quantify the full range of changes in health effects that will result from changes in the levels of ambient air pollution.

The very young, older adults, and persons with pre-existing respiratory and cardio-vascular disease are among those thought to be most strongly affected by air pollution. Airway inflammation induced by ozone and PM is especially problematic for children and adults with asthma, as it makes them more susceptible to having asthma attacks, consistent with recent asthma camp results (Thurston, *et al.*, 1997). For example, controlled human studies (e.g., Molfino *et al.*, 1991) have indicated that prior exposure to ozone enhances the reactivity of asthmatics to aeroallergens, such as pollens, which can trigger asthma attacks. In addition, ozone increases inflammation and diminishes immune function the lungs. This can make older adults more susceptible to pneumonia, a major cause of illness and death in this group.



Table 1. **Scope of health effects**

<b>Human Health Effects of Air Pollution</b>	
<b>Quantifiable Health Effects</b>	<b>Non-quantified/Suspected Health Effects</b>
Mortality*	Neonatal and post-neonatal mortality
Bronchitis - chronic and acute	Neonatal and post-neonatal morbidity
Asthma attacks	New asthma cases
Respiratory hospital admissions	Fetus/child developmental effects
Cardiovascular hospital admissions	Non-bronchitis chronic respiratory diseases
Emergency room visits for asthma	Cancer (e.g., lung)
Lower respiratory illness	Behavioral effects (e.g., learning disabilities)
Upper respiratory illness	Neurological disorders
Shortness of breath	Respiratory cell damage
Respiratory symptoms	Decreased time to onset of angina
Minor restricted activity days	Morphological changes in the lung
All restricted activity days	Altered host defense mechanisms
Days of work loss	(e.g., increased susceptibility to respiratory infection)
Moderate or worse asthma status	Increased airway responsiveness to stimuli
	Exacerbation of allergies

Source: Adapted from: The Benefits and Costs of the Clean Air Act 1990 to 2010, U.S. EPA, EPA-410-R-99-001 1999).

It is well established that both asthma mortality and asthma hospital admissions increased during the 1980's in the U.S. and other developed nations (e.g., Buist and Vollmer, 1990; Taylor and Newacheck, 1992). The highest rates are associated with inner city residence (Carr et al, 1992; Weiss and Wagener, 1990), and Latino or African-American origin (Carter-Pokras and Gergen, 1993; Coultas et al, 1993; Weiss and Wagener, 1990). Conventional air pollutants do not appear to be the driving force behind the global increase in the prevalence of asthma, since levels of conventional pollutants have generally declined while the asthma prevalence has increased. However, because more persons are acquiring asthma, an ever increasing number in the population will be affected by the aggravating effects of air pollution, especially among the economically disadvantaged. In many different regions of the world, studies have found a higher rate of hospitalizations for asthma in more polluted zones. For example, the chance of being hospitalized with asthma on days following high air pollution in New York City is much greater than in the surrounding suburbs, probably because the city has a much larger minority and low-income population (Thurston *et al.*, 1992, Saldiva *et al.*, 1996). In the words of Weiss and Wagener (1990), "whatever the reason for the increases, both asthma mortality and hospitalization continue to affect non-whites, urban areas, and the poor disproportionately. For example, the hospitalization rate for asthma is higher in New York City than anywhere else in the U.S. (Carr *et al.*, 1992).

Among the factors that may increase the susceptibility of residents of low-income urban neighborhoods to air pollution are:

- i. Enhanced individual susceptibility of minority populations to pollution effects (i.e., more compromised health status, due to genetic predisposition, impaired nutritional status, or severity of underlying disease);

- ii. Exposures to atmospheric pollution in center cities may be greater than those of the general population, due to higher local pollutant emissions combined with a lower prevalence of protective air conditioning;
- iii. Potentially enhanced exposures to a variety of residential risk co-factors such as cockroaches, dust mites, and indoor pollution sources (e.g., gas stoves used for space heating purposes); and/or
- iv. A greater likelihood of poverty, which is associated with reduced access to routine preventive health care, medication, and health insurance.

A number of studies suggest that the very young represent an especially susceptible sub-population, although the precise magnitude of the effects of specific levels of air pollution can be expected to vary with other underlying conditions. Lave and Seskin (1977) found mortality among those 0-14 years of age to be significantly associated with levels of total suspended particulates (TSP). More recently, Bobak and Leon (1992) studied neonatal (ages less than one month) and post-neonatal mortality (ages 1-12 months) in the Czech Republic, finding significant and robust associations between post-neonatal mortality and  $PM_{10}$ , even after considering other pollutants. Post-neonatal respiratory mortality showed highly significant associations for all pollutants considered, but only  $PM_{10}$  remained significant in simultaneous regressions. Woodruff *et al.* (1997) used cross-sectional methods to follow-up on the reported post-neonatal mortality association with outdoor  $PM_{10}$  pollution in a U.S. population. This study involved an analysis of a cohort consisting of approximately 4 million infants born between 1989 and 1991 in 86 U.S. metropolitan statistical areas (MSAs). After adjustment for other covariates, the odds ratio (OR) and 95% confidence intervals for total post-neonatal mortality for the high exposure versus the low exposure group was 1.10 (CI=1.04-1.16). In normal birth weight infants, high  $PM_{10}$  exposure was associated with mortality for respiratory causes (OR = 1.40, CI=1.05-1.85) and also with sudden infant death syndrome (OR = 1.26, CI=1.14-1.39). Among low birth weight babies, high  $PM_{10}$  exposure was associated, but not significantly, with mortality from respiratory causes (OR = 1.18, CI=0.86-1.61).

This study was recently corroborated by a more elegant follow-up study by Bobak and Leon (1999), who conducted a matched population-based case-control study covering all births registered in the Czech Republic from 1989 to 1991 that were linked to death records. They used conditional logistic regression to estimate the effects of suspended particles, sulfur dioxide, and nitrogen oxides on risk of death in the neonatal and post-neonatal period, controlling for maternal socioeconomic status and birth weight, birth length, and gestational age. The effects of all pollutants were strongest in the post-neonatal period and were specific for respiratory causes. Only particulate matter showed a consistent association when all pollutants were entered in one model. Thus, populations with relatively high percentages of children may have higher air pollution effects than in populations in study areas with fewer children (e.g., studies conducted more developed nations).

## 2.2 *Scope for valuation*

The literature reviewed above indicates that a wide variety of possible health endpoints could be affected ancillary to climate change mitigation policies. Some of these have not been quantified, and some of the effects listed as non-quantified may be related physiological expressions of those that are quantified. Of the quantified endpoints, Table 2 provides information about whether they have been (or could be) monetized, based on our understanding of the literature. We also consider the status of monetization for some of the non-quantified health effects. For each of these effects, we list the techniques used to provide monetary values. Willingness to Pay studies (WTP) are those that provide estimates of preferences for improved health that meet the theoretical requirements of neoclassical welfare economics. Cost of Illness (COI) is a technique that involves totaling up medical and other out of pocket expenditures. “Consensus” refers to the way in which these values were determined. They do not have much of an evidentiary basis. Each of these approaches and endpoints are discussed in more detail in Section 6.

The valuation literature for neonatal mortality, children, and cancer is very limited at the present time, but there is much interest in better valuing these endpoints in the U.S. Cancer valuation is discussed in Section 6. Valuing reduced mortality risks for newborns or children is one of the most challenging tasks in the field because children are generally not the key decision makers over their own health and safety. They are part of a family unit that makes choices for them. For this reason, some economists are devising household production function models to address these valuation questions (Tolley, 1999; Dickie, 1999) and devising novel strategies for measuring revealed preferences for improving child safety and health (such as examining the types of vehicles or bicycle helmets purchased). However, this literature has not yet matured.

In addition, we add here placeholders for two potentially important linkages that go from economic effects of GHG policies to health effects rather than the reverse. The linkage from unemployment to health refers to the possibilities that GHG policies might raise or lower unemployment rates from what they would otherwise be. A number of studies have linked unemployment to increased suicides, domestic violence, and alcohol and drug abuse. While this linkage is quite controversial in the health benefits literature, it has the potential to introduce a large, new set of ancillary benefits (or costs) to such studies. The second linkage — from lower incomes to lower health status — is of a similar type. GHG policies may lower incomes below BAU levels. A number of studies have shown how income is positively correlated with health status. Thus, if incomes fall or do not rise as fast as a result of a GHG policy, health may worsen absolutely or relative to what it would have been in the absence of a GHG policy. Again, this linkage is quite controversial, and is included here because of its potential to change thinking on the ancillary benefits/costs issue.

Table 2. Status of valuation of quantified and suspected health endpoints

Health Effects	Valuation Estimates Available?	Basis
<b>QUANTIFIED EFFECTS</b>		
Mortality: Adults	Y	WTP (caveats)
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	Y	Consensus
RAD	Y	Consensus
WLD	Y	Wage
Asthma Day	Y	WTP
Change in asthma status	N	
<b>NON-QUANTIFIED/SUSPECTED EFFECTS</b>		
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
<b>LINKS FROM ECONOMIC EFFECTS TO HEALTH</b>		
Health effects linked to unemployment	?	?
Health effects linked to lower incomes	?	?

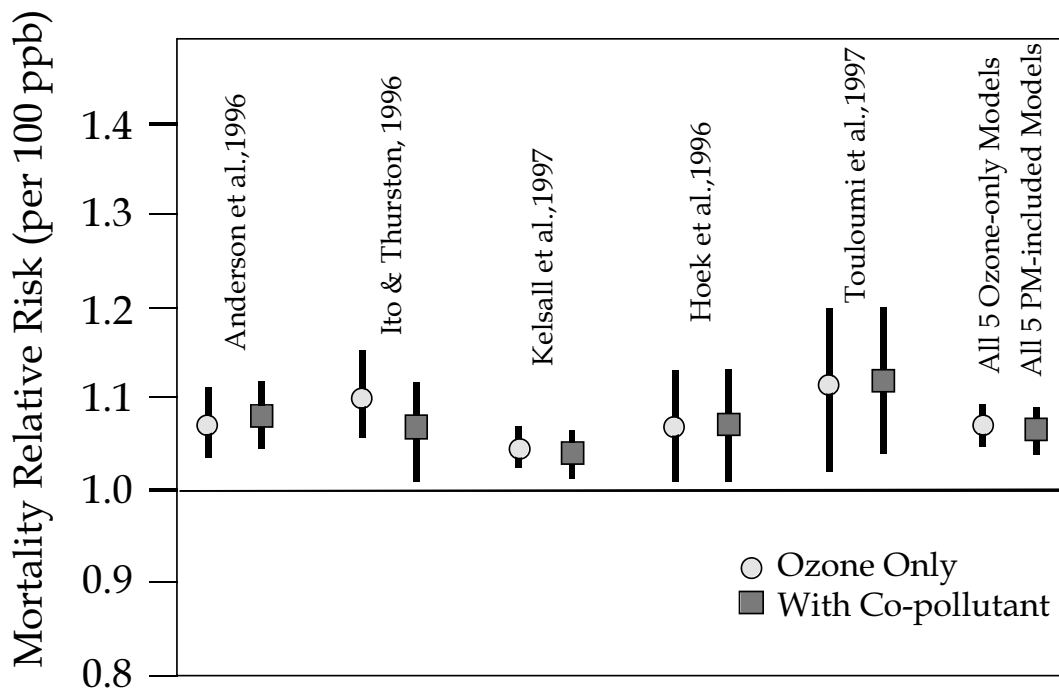
### 3. Scale of health effects

Observational epidemiology studies - both long-term and acute exposure studies - have shown compelling and remarkably consistent evidence of air pollution associations with adverse human effects, including: decreased lung function, more frequent respiratory symptoms, increased numbers of asthma attacks, more frequent emergency department visits, additional hospital admissions, and increased numbers of daily deaths. While not all of these impacts represent major financial impacts on the society, they often involve decreases in quality of life, and also provide additional weight of evidence of air pollution associations with the most serious effects, such as mortality.

In studies to date, the health effects associated with particulate matter (PM) and ozone (O<sub>3</sub>) have consistently been found. Particulate matter consists of two types of particles: primary and secondary particles. Primary particles come directly out of the tailpipes and stacks, and those are primarily carbonaceous particles. Secondary particles are formed in the atmosphere from gaseous pollutants, such as sulfates from sulfur dioxide and nitrates from nitrogen oxides (NO<sub>x</sub>). Ozone (O<sub>3</sub>) is an invisible irritant gas that is formed in the atmosphere in the presence of sunlight and other air pollutants. These ozone precursor pollutants, nitrogen oxides and hydrocarbons, as well as SO<sub>2</sub>, come from a variety of sources, including diesel automobiles (diesel contains more sulfur than gasoline), power plants, and industry.

As shown in Figures 2-5, there have been numerous studies published in recent years indicating consistent associations between both PM and O<sub>3</sub> exposures and both daily mortality and morbidity. It is interesting to note that the inclusion of PM along with ozone in the model has little effect on the O<sub>3</sub>-mortality effect, indicating that these associations are likely to be largely independent of one another.

Figure 2. Ozone mortality effect estimates across localities, with and without simultaneous inclusion of PM



Source: Thurston and Ito, 1999.

Relative Risk (RR) of Death for a 100ug/m<sup>3</sup> Increase in PM10

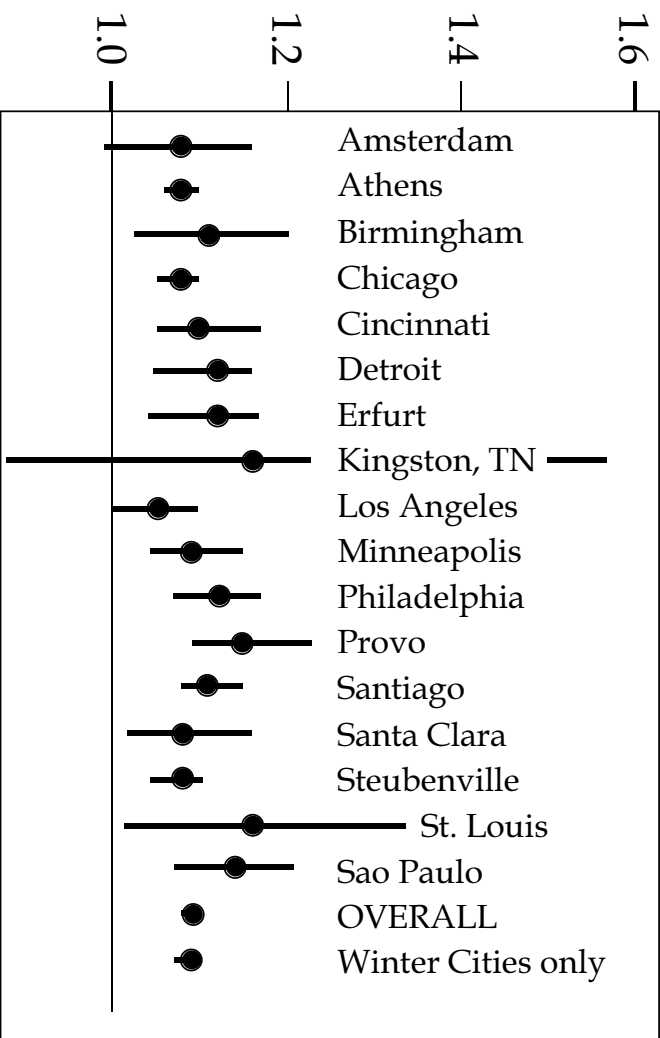


Figure 3. PM mortality effect estimates across localities

Source: Schwartz, 1997.

Relative Risk (RR) of Respiratory Hospital Admissions for a 100 ug/m<sup>3</sup> Increase in Ozone

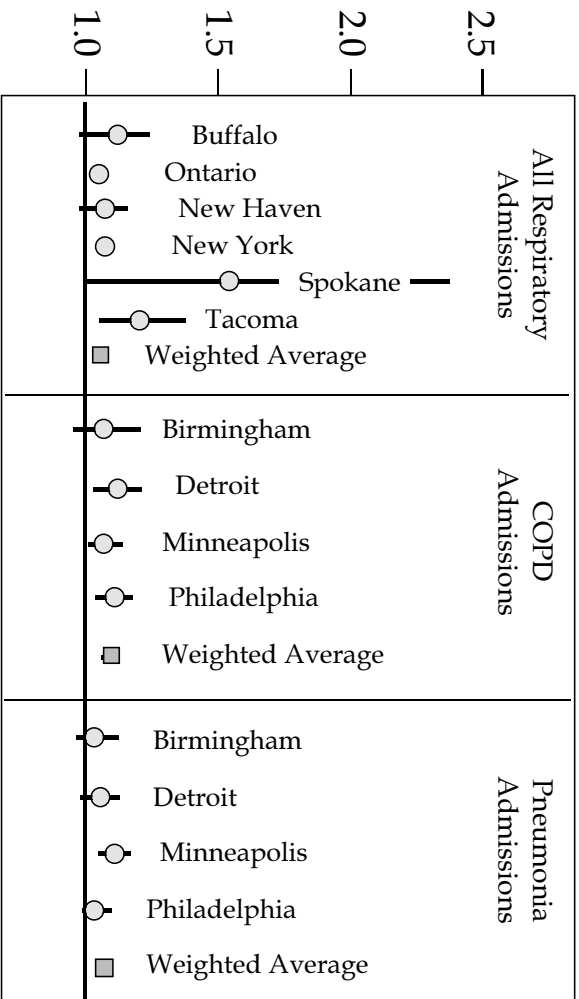
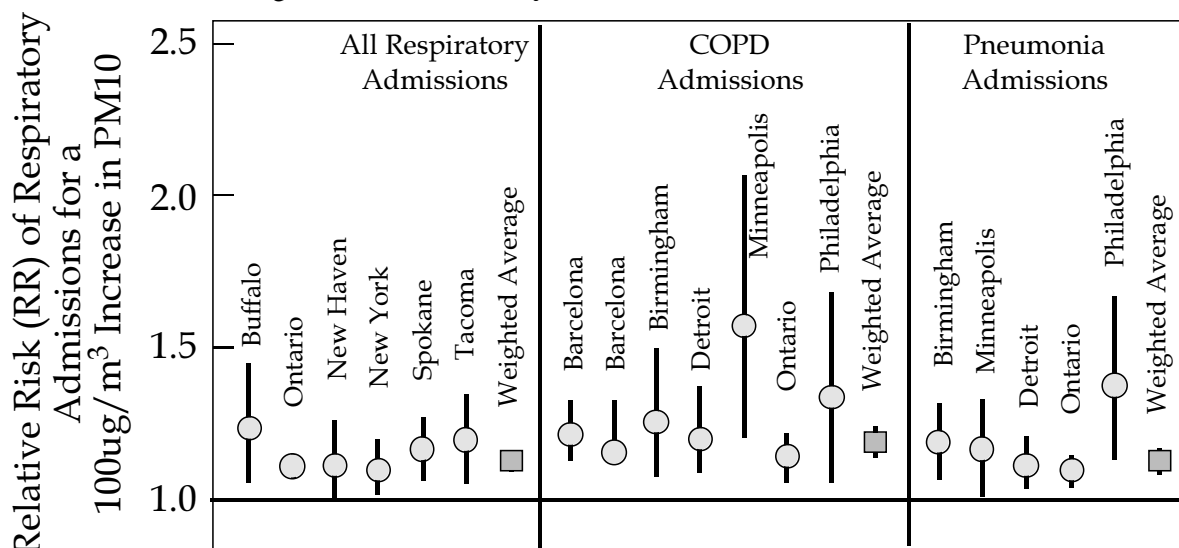


Figure 4. Ozone morbidity effect estimates across localities

Source: Schwartz, 1997.

Figure 5. **PM morbidity effect estimates across localities**



Source: Schwartz, 1997.

Three recent studies: the Harvard Six Cities study (Dockery *et al.*, 1993) the American Cancer Society (ACS) study (Pope III *et al.*, 1995) and the Health Effects Institute reassessment of other studies (Samet *et al.*, 2000), have shown significant associations between long-term exposure to air pollution and human mortality. This work confirms earlier cross-sectional results (e.g., Lave and Seskin, 1977, Ozkaynak and Thurston, 1988). While the magnitude of the chronic effects of air pollution is not definitively known, cohort and cross-sectional studies suggest that air pollution effects are significant and may cause as many as 30 000-60 000 deaths annually in the United States alone. As a percentage of death rates, a 10 ug/m<sup>3</sup> reduction in average daily levels of PM<sub>10</sub> has been found to reduce mortality by 0.5 per cent (Samet *et al.*, 2000) to as much as 7 per cent (Dockery *et al.*, 1993). This range is a product of a variety of factors, including varying research methods, as well as for unexplained reasons, but the major difference is due to the fact that the lower estimates include only the effects of day-to-day acute air pollution exposure effects, while the higher estimates also include cumulative effects associated with longer-term (i.e., lifetime) air pollution exposures. For example, U.S. death rates over the last decade were about 800 per 100 000, thus a 10 ug/m<sup>3</sup> reduction (say from an average daily mean of 75 ug/m<sup>3</sup>) could result in 4 to 56 fewer premature deaths annually per 100 000 people. Calculations in Davis *et al.* (1997) found that controlling air pollution from fossil fuel combustion in transportation, residential use, energy, and industry in order to reduce greenhouse gas emissions might reduce as many deaths in the U.S. each year as currently occur due to traffic crashes or HIV. Globally, reductions in air pollution tied with mitigating greenhouse gases could reduce 4.4-11.9 million excess deaths by 2020. Additional burdens on public health from morbidity associated with these exposures would be expected to be diminished as well, although these are not readily calculated on a global scale.

Table 3 (from a study recently conducted in Chile) shows examples of the types of C-R coefficients that can be derived from the epidemiological literature for an analysis of the public health implications of GHG mitigation (Cifuentes *et al.*, 1999). It is notable that the index pollutant used in each category is particulate matter (either as PM<sub>2.5</sub> or PM<sub>10</sub>). This is likely due to the fact that, of the air pollutants, most of the most serious effects (e.g., mortality) are found for PM, rather than for the gaseous air pollutants. Therefore, this is the “index” of air pollution effects most commonly selected for such studies.

**Table 3. Summary of C-R coefficients (B’s) used in a public health analysis of Chile GHG mitigation co-benefits**

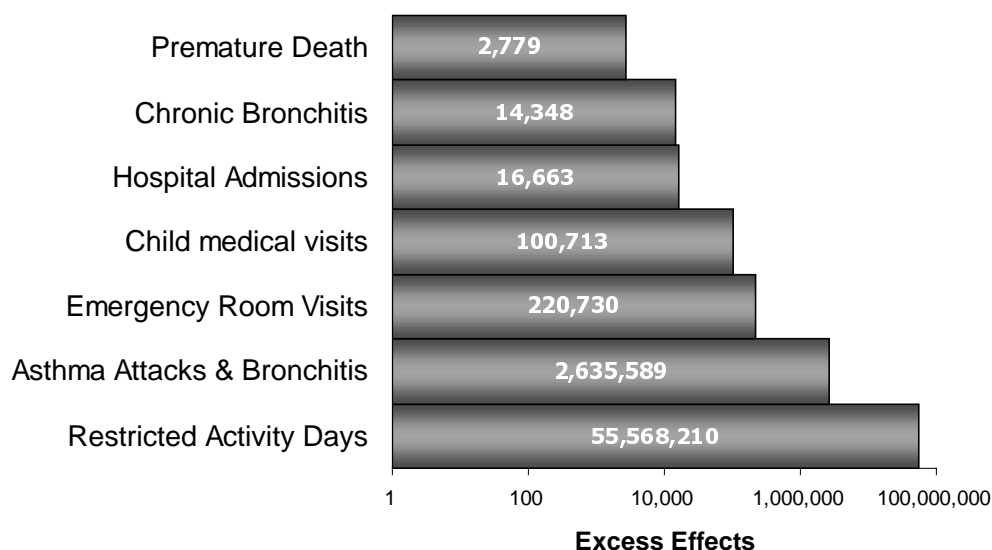
Endpoints	Age Group	Pollutant	Mean	t stat	Source
Chronic Mortality	All	PM2.5	0.00450		Ostro 1996 (high case)
Chronic Bronchitis	> 65 yrs	PM10	0.02100	4.2	Schwartz et al,1993
Premature Deaths (short term)	All	PM2.5	0.00120	3.9	Own analysis
Hospital Admissions RSP	> 65 yrs	PM10	0.00169	3.8	Pooled
Hospital Admissions COPD	> 65 yrs	PM10	0.00257	6.4	Pooled
Hosp. Adm Congestive heart failure	> 65 yrs	PM10	0.00098	3.2	Schwartz & Morris, 1995
Hosp Adm Ischemic heart failure	> 65 yrs	PM10	0.00056	2.7	Schwartz & Morris, 1995
Hospital Admissions Pneumonia	> 65 yrs	PM10	0.00134	5.1	Pooled
Asthma Attacks	All	PM10	0.00144	4.6	Ostro et al, 1996
Acute Bronchitis	Childs	PM2.5	0.00440	2.0	Dockery et al., 1989
Child Medical Visits LRS	Childs	PM10	0.00083	2.5	Ostro et al, 1999
Emergency Room Visits	All	PM10	0.00222	5.2	Sunyer et al, 1993
Shortness of Breath (days)	Childs	PM10	0.00841	2.3	Ostro et al, 1995
Work loss days (WLDs)	18-64 yrs	PM2.5	0.00464	13.2	Ostro et al, 1987
RADs	18-64 yrs	PM2.5	0.00475	16.5	Ostro et al, 1987
MRADs	18-64 yrs	PM2.5	0.00741	10.5	Ostro et al, 1989

Source: Cifuentes et al, 1999).

The same Chilean analysis in Figure 6 shows an example of the results of applying these C-R functions to estimate the effects avoided via GHG mitigation. These public health benefits are associated with reductions in PM<sub>2.5</sub> of 10% below what would occur in the year 2020 without any GHG mitigation. It is notable that there are thousands of deaths estimated as potentially avoidable by the co-pollutant reductions also achieved by GHG mitigation.



Figure 6. **Total estimated effects avoided in Chile during the period 2000 to 2020**



Source: Cifuentes, et al., 1999.

Most of the available air pollution studies used in such quantifications of air pollution effects have been conducted in developed countries. A series of recent studies indicate that assessing the coefficients of mortality tied with various pollutants may not completely reflect the relative importance of pollutants in all countries. For any society, deaths that occur at earlier ages are deemed more important than those that occur later in life, as they result in more years of life lost. For instance, one study in Delhi, India, found that children under 5 and adults over age 65 were not determined to be at mortality risk from air pollution. Perhaps this was because other causes of death (notably infectious diseases) predominated in those who survive to reach these age groups, or because of inadequate power to detect such effects (Cropper et al, 1997). However, persons between the ages of 15-45 were found to be at increased risk of death from air pollution. The reason that the Relative Risk (RR) estimates from this study are lower than found in Philadelphia is not known. This may be due to differences in pollution mix (e.g., PM composition), but it could also be due to a higher infant mortality rate that removes a potentially susceptible sector from the adult population. Because the population distribution in India includes many more persons, the net impact on the country from air pollution measured in terms of years of life lost is considerable. Although the percentage increase is lower in India than in more developed nations, the cumulative effect (i.e., attributable risk) in India is as high (Table 4). Thus, the mortality effects of air pollution in developing nations can be an important factor in GHG considerations.

Table 4. **Percentage increase in mortality and years of life lost per 100 ug/m3 increase in TSP: Delhi vs. Philadelphia<sup>1</sup>**

Mortality End Point	Delhi (% change per 100 ug/m3)	Philadelphia (% Change per 100 ug/m3)	Years of Life Lost	
			Delhi	Philadelphia
By selected cause				
Total deaths				
CVD	2.3	6.7	51,403	51,108
Respiratory	4.3	9.2		
	3.1	10.2 (Pneumonia) 17.8 (COPD)		

Source: <sup>1</sup> Cropper et al, "The Health Benefits of Air Pollution Control in Delhi," Amer. J.Agr. Econ. 79 (5, 1997): 1625-29.

Similar assessments can be made with respect to morbidity and disability tied with a "base," or reference case. Disability years of life lost can be estimated by considering the average age at which specific disabilities known to be worsened by air pollution, such as asthma and other chronic lung diseases, occur. This then provides a baseline against which to estimate likely reductions of direct labor productivity, or some other measure of output, associated with this disability. WHO has derived estimates of DALY (disability adjusted life years) from various major causes for 1990 and future years (Murray and Lopez, 1996). Respiratory diseases, which can be associated with air pollution, rank within the top 5 causes of deaths worldwide for adults and were the leading causes of deaths in children (WHO, 1996).

#### 4. Credibility of health effects linkages

Uncertainties characterize many parts of the process by which information is obtained on the air pollution-public health consequences of GHG mitigation policies. Among the important issues that require clarification are:

- the credibility of dose-response relationships between specific pollutants and specific types of mortality;
- the transferability of coefficients of these relationships derived in more developed countries to less developed ones;
- the extent to which morbidity can be quantified for specific pollutants and mixtures of pollutants and the reliability of this quantification;

- the mix of chronic and acute impacts for which quantification can and should be undertaken;
- the credibility of methods for expressing preferences for health improvements in money (or other) terms.

For any given health outcome, the size of the estimated health risk is often at issue and is usually expressed as the increase in risk per increase unit of exposure. While some monitors exist in many cities of the world, there remain serious problems with exposure misclassification. Where models are employed to estimate exposure, these are also quite limited by gaps in the availability of model inputs, such as accurate emissions source data. Recent assessments also indicate that the relative importance of indoor and outdoor air varies substantially in developed and developing countries (Wang and Smith, 2000). The use of experimental models and animal studies for the estimation of human risks is also a point of contention. Where they exist, epidemiological studies are usually relied upon for RR estimates.

Although air pollution, and especially PM air pollution, has been compellingly linked with excess mortality by a substantial body of epidemiological research, there are aspects of this association that are still uncertain. There is always concern that some confounder, another variable not adjusted for in the analysis but correlated with the exposure and causally related to the effect, might actually be responsible for an association found by an epidemiological study. This is especially of concern when there is no known biological plausibility of the effects noted. However, in the case of air pollution, the effects are certainly biologically plausible, based on controlled studies and past documented episodes, such as the London Fog of 1952. In addition, health effects from air pollution have been documented at so many locales and in so many different populations. The main question today is whether previously documented mortality effects can be experienced at more routine ambient levels. Other uncertainties in these analyses include the following issues:

#### 4.1 Causality

Because the epidemiological studies alone cannot definitively prove causation, other evidence should be brought to bear, such as toxicological studies. For example, clinical studies have demonstrated decreased lung function, increased frequencies of respiratory symptoms, heightened airway hyper-responsiveness, and cellular and biochemical evidence of lung inflammation in exercising adults exposed to ozone concentrations at exposures as low as 80 parts per billion for 6.6 hours (e.g., Folinsbee *et al.*, 1988, and Devlin *et al.*, 1991). Airway inflammation in the lung is among the serious effects that have been demonstrated by controlled human studies of air pollution at ambient levels. Airway inflammation is especially a problem for children and adults with asthma, as it makes them more susceptible to having asthma attacks. For example, recent controlled human studies have indicated that prior exposure to ozone can enhance the reactivity of asthmatics to aeroallergens, such as pollens, which can trigger asthma attacks (Molfino *et al.*, 1991). Thus, air pollution exposures may indirectly exert their greatest effects on persons with asthma, by increasing inflammation in the lung, which then heightens the responsiveness of asthmatics' lungs to all other environmental agents that may cause an exacerbation. The precise mechanism(s) by which air pollution could cause the pollution-mortality associations indicated by epidemiologic studies are not fully understood. The absence of understanding of mechanisms is also the case with respect to cigarette smoking and the many diseases known to be associated with it. For air pollution, the epidemiological results are supported by a large body of data from controlled exposure studies in animals and humans that consistently demonstrate pathways by which pollution can damage the human body when it is breathed (e.g., US EPA, PM Criteria Document, 1999).

## 4.2 *Other pollutants*

PM concentrations are often correlated spatially and over time with the concentrations of other ambient pollutants, making it more difficult to unambiguously separate the effects of the individual pollutants. Thus, it is unclear how much each pollutant may individually influence elevated mortality rates. As a result, most cost-benefit studies choose one index air pollutant, rather than estimating effects for multiple air pollutants individually and then adding their effects to get a total air pollution effect. This focus on a single pollutant provides a conservative approach to estimating air pollution effects. In fact, several recent analyses (e.g., Thurston and Ito, 1999) suggest that ozone and PM air pollution effects are relatively independent, since controlling for one pollutant has only modest effects on the relative risk (i.e., the C-R function) of the other (see Figure 2). Thus, the use of a single index pollutant in Cost-Benefit Analyses underestimates the overall public health effects and monetary valuations of air pollution changes. However, this use of a single pollutant may overestimate the effect assigned to a particular pollutant if it is only serving as an index of effects actually caused by the overall mixture of pollutants.

## 4.3 *Differences between central site and personal pollution exposures*

It has been pointed out that central site monitoring station measurements are poorly related to the personal exposures of individuals, which introduce some error into the air pollution-health effects relationship. While this is true on an individual level, it is not always so on an aggregate (i.e., population average) level, as shown by Mage and Buckley, 1995). In making estimates of the personal exposure of individuals to outdoor air pollution (the type of air pollution under consideration for change), the measurements from a central site monitoring station may actually be more appropriate than measurements made indoors, or even than personal samplers collecting all particles in the air, including those of indoor origins. Recent studies (e.g., Leaderer *et al.*, 1999) show significant correlations between central site monitors of  $PM_{2.5}$  and the ambient concentrations of  $PM_{2.5}$  outside the homes of individual study participants. Moreover, fine particles of outdoor origins permeate readily to the indoors. This is exemplified by sulfates, a common outdoor fine particle component that is usually not confounded by indoor sources, and which showed high correlations ( $r=0.88$ ) between outdoor measurements and personal exposures during the P-Team study (Ozkaynak *et al.*, 1996). Conversely, total personal  $PM_{2.5}$  exposure measurements can be greatly influenced by indoor sources (e.g., due to dusts, molds, side-stream cigarette smoke, etc.), but these are not the focus of government regulations or of the health effects epidemiology used to develop C-R functions.

If personal samples of  $PM_{2.5}$  were in fact collected and used in an epidemiologic study of the mortality effects of  $PM_{2.5}$  of outdoor origins, for example, the exposure data would be confounded by the PM contribution from indoor sources, which would then need to be separated out prior to the epidemiologic analyses of the effects of outdoor air pollution on health. Central site measurements are correlated with community outdoor concentrations. Therefore, community-based measurements are also correlated with the contribution of outdoor air pollution to indoor exposures. The use of outdoor measurements actually avoids potential confounding by indoor PM pollution.

#### **4.4**      *Shape of the C-R function*

The shape of the true air pollution C-R function is uncertain. Most recent analyses assumes the C-R function to have a log-linear form throughout the relevant range of exposures. If this is not the correct form, or if certain scenarios predict concentrations well above the range of concentration under which the C-R function developed, then avoided public health effects may be miss-estimated. The choice of whether to assume that there is a threshold of effects will also influence the effect estimates. The existence of a level below which there are absolutely no effects of air pollution exposure has not been documented in the literature, so there is at this time no definitive basis upon which to set a threshold of effects. However, alternative threshold assumptions can be investigated as part of a sensitivity analysis.

#### **4.5**      *Regional and country-to-country characteristic differences*

While air pollution epidemiology studies indicate a general coherence of results, variability exists in the quantitative results of different air pollution studies (i.e., the RR's per  $\mu\text{g}/\text{m}^3$  of pollution). This variability may in part reflect region-specific C-R functions resulting from regional differences in potentially important factors such as socio-economic conditions, age-distribution of the population, population racial composition, and health care practices. If true regional differences exist, applying these C-R functions to regions other than the study location could result in miss-estimation of effects in these regions. The scientific literature does not presently allow for a region-specific estimation of health benefits in many areas. However, the use of relative risks based upon local rates of disease (rather than absolute numbers of effects per  $\mu\text{g}/\text{m}^3$  of pollution) could minimize miss-estimation resulting from the use of uniform C-R functions across localities.

#### **4.6**      *PM composition*

In the case of PM pollution, factors such as the physical and chemical composition of particles in the ambient air can be expected to vary its toxicity, which would change the C-R function for this key pollutant. Some things, however, can be tentatively concluded about the relative toxicity of different PM from various types of sources. For example, an analysis of 1980 cross-sectional variations in mortality across the U.S. by Ozkaynak and Thurston (1987) found that coal-combustion and industrial process-derived particles were more strongly associated with mortality than were oil combustion, gasoline auto emissions, or soil-derived (i.e., wind blown) particles. In addition, recent analyses indicate that fine particles (i.e.,  $d_a < 2.5 \mu\text{m}$ ) are more strongly associated with adverse health effects than are coarser particles (i.e.,  $d_a < 2.5 \mu\text{m}$ ) (e.g., Thurston *et al.*, 1994).

Recent studies also suggest that diesel emissions may contain especially toxic particles. Indeed, in the published literature, diesel exhaust particles have been associated with a worsening of respiratory problems. Recent epidemiological studies indicate that respiratory problems are worsened in residential areas closer to diesel truck traffic (de Hartog *et al.*, 1997) (Brunekreef, *et al.*, 1997a). Furthermore, clinical studies confirm that diesel particles can increase asthma responsiveness by causing a skewing of the immune response towards increased Immunoglobulin E (IgE) production, in turn causing allergic inflammation. Recent experimental studies in animals and humans have shown that diesel particulate, through its effects on cytokine and chemokine production, which are known to be associated with an increased inflammatory response, enhances this IgE production (e.g., see: Nel *et al.*, 1998). Thus, exposure to diesel air pollution may well act by increasing an asthma patient's general responsiveness to any and all allergens and pollens to which they are already allergic. This would increase the chance that acute asthma problems will be experienced in a given population of persons with asthma.

Thus, PM from different emission source categories may have different health implications. This would suggest that the health consequences of GHG mitigation are sensitive to how GHGs are reduced. Technologies that resulted in more reductions of fine particulates from coal combustion, industrial processes, or diesel fuel combustion would result in greater health benefits.

#### **4.7      *Exposure/Mortality lags***

It is presently not known whether there is a delay between changes in air pollution exposures and changes in health effects (e.g., mortality rates) in chronic (long-term) air pollution-health effects relationships. This is not a concern in acute studies, however. The existence of such a delay could be important for the estimation of health and monetary benefits. Although there is no specific scientific evidence of the existence of an effects lag, current evidence on adverse health effects of smoking suggests that effects might well be delayed a matter of years (U.S. DHHS, 1990). If this is shown to also be the case for air pollution, a lag structure of the non-acute portion of the air pollution effects could be incorporated into a Cost-Benefit analysis.

#### **4.8      *Cumulative effects***

Cross-sectional and long-term cohort studies (e.g., Pope *et al.*, 1995) are thought to relate primarily to PM-associated cumulative damage, as they are larger than the short-term mortality estimates reported from time-series studies. However, the relative roles of acute and chronic air pollution exposures are not defined at this time, and it is not yet possible to separate them out in Cost-Benefit Analyses (e.g., by using differing life-shortening implications).

## 4.9 *Life-shortening*

The public health burden of mortality associated with exposure to ambient PM depends not only on the increased risk of death, but also on the length of life-shortening that is attributable to those deaths. The most recent U.S. EPA PM Criteria Document concluded that “Confident qualitative determination of years of life lost to ambient PM exposure is not yet possible; life-shortening may range from days to years.” (U.S. EPA, 1996). A new analysis has now provided a first estimate of the life-shortening associated with chronic PM exposure. Brunekreef (1997) reviewed the available evidence of the mortality effects of long-term exposure to particulate matter air pollution and, using life table methods, derived an estimate of the reduction in life expectancy that is associated with those effect estimates.

Based on the results of Pope et al (1995) and Dockery *et al.* (1993), a relative risk of 1.1 per 10  $\mu\text{g}/\text{m}^3$  exposure over 15 years was assumed for the effect of particulate matter air pollution on men 25-75 years of age. A difference of 1.11 years was found between the “exposed” and “clean air” cohorts’ overall life expectancy at age 25. Looked at another way, this would imply that the expectation of the life span of the persons who actually died from air pollution was reduced by more than 10 years, since they represent a small percentage of the entire cohort population.

The above study of air pollution’s effect on life expectancy considered only deaths among adults above 30 years of age, but deaths among children can logically have the greatest influence on a population’s overall life expectancy. As discussed above, some of the older cross-sectional studies and the more recent studies by Bobak and Leon (1992, 1999) and Woodruff *et al.* (1997) suggest that infants may be among the sub-populations that are especially affected by long-term PM exposure. Although it is difficult to quantify, any premature air pollution-associated mortality that occurs among children due to long-term pollution exposure, would significantly increase the overall population life-shortening. This is because all other estimates of life-shortening have been based solely on the impacts of long-term pollution exposures to adults 30 years and older. Therefore, considerable uncertainty remains as to the amount of life-shortening associated with long-term exposure to air pollution.

## 5. **Scale of values**

Much of the justification for environmental rulemaking rests on estimates of the benefits to society of health improvements. Within this set of effects, reductions in risk of death are usually the largest category of benefits, both within the health category and compared to other categories. For instance, mortality risk reductions are arguably the most important benefit underlying many of the USEPA’s legislative mandates, including the Safe Drinking Water Act, CERCLA, the Resource Conservation and Recovery Act and the Clean Air Act. In the most recent analysis of the benefits and costs of air quality legislation, *The Benefits and Cost of the Clean Air Act Amendments of 1990* (USEPA, 1999), reductions in premature mortality are valued at \$100 billion annually out of \$120 billion in total benefits, compared to costs of about \$20 billion. Even though health costs/benefits of air pollution changes are harder to identify and quantify than those for control costs, European and Canadian studies similarly find that mortality risk reductions dominate any analysis of pollution reductions (EXTERNE, 1999; AQVM, 1999).

Next to reductions in mortality risk, reductions in the probability of developing a chronic respiratory disease have been estimated to be the most valued, recognizing that values for other types of diseases are sparse. Reductions in various acute effects are lower valued. It is recognized that reduced pollution may lead to decreased acute consequences of disease, such as an emergency room or a daily hospital visit. Reductions in these endpoints tend to be far more highly valued than the avoidance of a day of acute symptoms as defined in standard health benefits analysis, i.e., coughs, sneezes, and the like. Nevertheless, the former are based on medical costs, while the latter are based on willingness to pay (WTP)—the appropriate measure of preferences for health improvements (see Section 6).

Table 5 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 willingness to pay for reducing risks of mortality and chronic morbidity is expressed, for convenience, in terms of the value of a statistical life (VSL) and the value of a statistical case of chronic disease (VSC). It is important to note that this term is merely a shorthand for the WTP for a given risk reduction divided by that risk reduction. This relationship is convenient because the VSLs or VSCs can be multiplied by estimates of the “lives saved” or “chronic cases saved” to obtain benefits.

The table shows quite close agreement on the size of the best or midpoint VSLs and VSCs. 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. 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. In other analyses, EPA and TAF have done the same thing (see Section 6 for more information).<sup>35</sup> 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 Markandya et al, 1996). 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.

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<sup>35</sup> Other adjustments to VSLs have been made for latency (ExternE), for health status (basically the “harvesting” issue) (Markandya, 1999) and for a range of attributes, such as dread and voluntariness (US EPA, 2000).



Table 5. Comparison of unit values used in several major studies or models. (\$1990).

Values	US EPA <sup>a</sup>			US TAF <sup>b</sup>			Canada AQVM <sup>c</sup>			Europe ExternE <sup>d</sup>
	<i>Low</i>	<i>Central</i>	<i>High</i>	<i>Low</i>	<i>Central</i>	<i>High</i>	<i>Low</i>	<i>Central</i>	<i>High</i>	<i>Central</i>
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 Costs and Benefits of the Clean Air Act Amendments of 1990. 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. Air Quality Valuation Model Documentation, Stratus Consulting for Health Canada. Low, central, and high estimates are given respective probabilities of 33%, 34%, and 33%.
- c. Tracking and Analysis Framework, developed by a consortium of U.S. institutions, including RFF. Low and high estimates are the 5% and 95% tails of the distribution.
- d. ExternE report, 1999. Uncertainty bounds are set by dividing (low) and multiplying (high) the mean by the geometric standard deviation (2).

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<sup>th</sup> and 95<sup>th</sup> per centile. Error bounds in the latest EXTERNE report are established as one half (low) and twice (high) the geometric mean.

It is worth noting that 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 willingness-to-pay 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 cost of illness 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 (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.

## **6. Credibility of valuation estimates**

So far, we have presented some idea of the range of “best” estimates for avoiding health effects or reducing their risks as taken from a variety of major studies or models in developed countries. Now, we describe their uncertainties and assess the state of the information available for making these valuations.

In this section, we first delve into the details of the various valuation estimation procedures and their limitations, tackling mortality risk valuation first and then chronic and acute morbidity valuation. At the end of each of these subsections, tables are presented that summarize the state of the literature. We then address the issue of valuation in a developing country context, examining the literature on benefits transfers and indigenous valuation.

There are obviously many ways to make such assessments. This assessment is only intended to provide a comparison among the various estimation approaches for each endpoint, rather than an absolute assessment of accuracy. We use three criteria for making such a judgement: (i) the degree to which methods of estimation are based on preferences for such health improvements (which we take to be synonymous with following welfare economics principles); (ii) the number of studies that have followed this technique (being an imperfect measure of degree of consensus and attractiveness of the technique to researchers); and (iii) additional major limitations of the technique (to capture other limitations of the approach, such as data limitations). Based on our subjective judgement, we then provide a rating of the reliability of the different approaches to estimating the value for the endpoint in question: A (very reliable) to D (unreliable)

## 6.1 *Concepts for valuing health*

Underlying any attempt to attach an economic value to health effects is the idea that individuals have preferences that extend over environmental quality and its implications as well as over market goods (and other non-market goods besides environmental goods). If this assumption is accepted then, *in principle*, it is possible to deduce how individuals trade off environmental quality or their health against other services they value. This assessment can be made by attempting to measure how much in the way of other services individuals are willing to give up in order to enjoy health benefits.<sup>36</sup> The expression of these values in money terms is just a convenient shorthand for what people are willing to give up in alternative real consumption opportunities. In the aggregate, this measure is given by the sum of individuals combined *willingness to pay* for the specified improvement.

## 6.2 *Mortality risk valuation*

### 6.2.1 *Evaluating estimation approaches*

We have identified five approaches to estimating preferences for reducing mortality risks and expressing these preferences in monetary terms: the human capital approach, various revealed preference approaches (most importantly the hedonic labor market approach, but also the consumer products approach), and stated preference techniques that address health and that do not.

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, it 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 no longer used.

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The notion that such individual tradeoffs fully describe society's interest in environmental quality is by no means universally accepted, particularly among non-economists who are highly critical of economic valuation in general and benefit-cost analysis in particular. For an excellent summary of the economic argument see Freeman (1993). Even if one accepts that WTP is an acceptable measure of individual valuation, distributional effects will complicate the effort. These complications arise because changes in environmental quality or health often will themselves change the real income (utility) distribution of society, taking into account both non-market and market benefits. It is important to recognize that a valuation procedure that adds up individual WTP is not capturing individual preferences about changes in the income distribution, even though these clearly do matter from a policy perspective. This is a complicated issue that is beyond the scope of this paper to address.

The two most common approaches for 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. One study asks workers their perception of the death risks they face in order to address the issue of whether their behavior would be consistent with perceived risks rather than actual risks and that these two types of risks might diverge. These studies are numerous and form the foundation for most VSL estimates. However, they are problematic when applied to assessing the willingness to pay to avoid health effects of air or other pollutants. 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.<sup>37</sup> 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 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 per cent, then *discounted* remaining life expectancy is only 16 years, and the value per life-year saved rises to approximately \$300 000.<sup>38</sup>

Markandya et al have recently developed another, relatively ad hoc approach to adjusting VSLs for a variety of shortcomings. The elaborate set of adjustments to the standard VSL (\$2.4m) 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 \$2.4m. The upper bound estimate is 70% of the VSL (\$1.7m), 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 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.

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<sup>37</sup> 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).

<sup>38</sup> 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.

There is also a small literature of consumer preference studies (generally and unfortunately ignored) 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) and in driving behavior under different speed limits (Greenstone and Ashenfelter, 1999). These studies tend to find lower VSLs. For instance, Greenstone and Ashenfelter find a VSL of \$1.56 million in \$1999. One problem with some of these studies is statistically separating the mortality risk-reducing attribute from other attributes of value to individuals.<sup>39</sup>

The stated preference approaches, of which contingent valuation and conjoint analysis are the two most prominent, are survey approaches that set up choice situations. These methods ask individuals to choose among various hypothetical choices. For instance, they ask whether individuals are willing to pay some amount, or to vote yes on a referendum, or to prefer one package of attributes over another, in order to acquire reductions in mortality risk. The ability of conjoint analysis to recover preferences is a matter of debate. 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 study used conjoint analysis to examine WTP for reduced mortality risks in a pollution-type context (Desvousges et al, 1998) 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 context applicable to mortality risk reductions from pollution: Johannesson and Johansson (1995) and 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 it offers respondents what is actually an unrealistically huge reduction in risk.

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<sup>39</sup>

We ignore here the large body of literature using an hedonic property value approach. His 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. See Smith (1999) for a recent example.

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). The WTP estimates from this contingent valuation survey imply values of a statistical life (VSLs) considerably lower than those in the standard literature. The estimates of mean WTP imply a VSL of approximately \$800 000 to \$2 million (US dollars).<sup>40</sup> They also find that annual WTP for a risk reduction of comparable size is significantly lower (\$200 000) when the risk reduction takes place later in life (at age 70, occurring over 10 years, instead of beginning now and taking place over the next ten years). Further, they find that age has a relatively minor effect on the VSL and that *physical* health status has no effect. However, mental health affects VSL, with more “mentally” healthy individuals being willing to pay more for a given risk reduction. An individual’s vision for their future health was found to also affect their current WTP for a future risk reduction.

There are two studies in the literature that estimate the WTP to reduce cancer mortality risk. In addition, EPA has produced an unpublished paper that adjusts the standard VSLs to account for unique elements of the cancer risk, such as dread. Magat, Viscusi and Huber (1996) use conjoint analysis to determine the tradeoff between the risk of dying in an auto accident and the risk of dying from terminal lymphoma. They found that risks traded at one-to-one, implying equal VSLs to that found in the auto-accident context. Hamilton, Viscusi, and Gayer (1999) conducted an hedonic property value study near a Superfund site to estimate the WTP to reduce risks from that site. They found VSLs around \$4.5 million. The EPA adjustment procedures lead to a range of one-half to three times the benefits using the “standard” U.S.EPA’s VSL of \$5.8 million (\$1999).

The willingness to pay measures will be theoretically superior to the “supply-side” measures of health damage that are usually used when WTP measures are unavailable. These measures include the value of productivity lost or expenditures on avoidance and amelioration (e.g., medical costs). WTP measures are preferred because they theoretically capture the complete value of such effects, including “pain and suffering.” Out of pocket medical expenditures, for instance, are likely to provide a lower bound on willingness to pay (particularly if medical costs are partly or completely paid by insurance or “the State”), since they do not include any of the intangible costs of reduced health, such as reductions in quality of life suffered by the ill person and any caregivers (just as the total expenditure on food is not a complete measure of the value people place on sustenance).<sup>41</sup>

Ideally, the out-of-pocket expenditure data can be supplemented or replaced by cost data that reflect the opportunity costs of resources on avoidance or amelioration. In a developing country, what data are available on costs may reflect wages and other inputs to medical services that are distorted through government policies, such as minimum wage laws. In such cases, cost data will not reflect social opportunity costs, and further adjustments are desirable.

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<sup>40</sup> These estimates are, however, in line with some revealed preference studies based on consumer behavior (Viscusi 1992; Greenstone and Ashenfelter 1999).

<sup>41</sup> Regulatory constraints may cause amelioration expenditures to overstate damages if very strict regulatory standards apply.

Part of the VSL literature has taken a very different approach than that discussed so far. Carrothers, Graham, and Evans (1999) use a QALY approach to estimate a VSL appropriate to the “short-term” or “time-series” air pollution-mortality studies. They both discount the life years lost because people who die prematurely are not in good health and assume that only a few “days to years” are lost because of exposure to air pollution. Then, they convert the QALY to a money metric using \$50 000 per QALY, a commonly accepted cost benchmark for medical interventions, although not necessarily the appropriate number in this air pollution context. The result is a VSL less than 10 per cent of that used by EPA (\$4.8 million, \$1990). Note that while the assumption of losing only a few days to years may be appropriate for the time-series studies, it does not hold up well for the long-term or prospective studies that follow people for eight years and record when they die. Note also that Krupnick et al (2000) find that poor physical health does not reduce WTP, which calls into question the discounting of future life years for ill health.

The authors also argue that pollution exposures may lead to morbidity effects that exceed those picked up by the time series studies. This could happen if the effects on risks of non-fatal heart attacks which later induces premature death add up to more life expectancy loss than those from direct fatalities. This speculation requires more research.

### 6.2.2 *How credible are the VSL estimates from these approaches?*

The ideal situation for reducing uncertainty about the use of the estimates from one or more of these approaches would be to have a set of unambiguous theoretical predictions borne out by a complete and unambiguous set of empirical findings. As Table 6 shows, the body of literature varies from this ideal.

Neoclassical welfare economics, in particular, the life-cycle utility model, lies at the heart of the theoretical modeling. Its predictions about the effects of various factors on WTP are provided in the first row of the table. WTP should clearly increase with the size of the risk change; indeed, subject to some minor caveats, the life cycle model predicts a proportional relationship. This implies that the VSL would be constant for any risk change. The model also implies that the further in time any risk change begins, the lower WTP should be. The effect on WTP of baseline risk has been very recently studied by Pratt and Zeckhauser, who show that those facing higher baseline risks should be willing to pay more for a given risk reduction (the “dead anyway” effect). Higher incomes or wealth should be related to higher WTP. The age effect varies depending on whether the individual can borrow against future earnings. With borrowing, the predicted relationship is an inverted U-shape, peaking, according to these studies, at around 40. Finally, the models do not make a prediction on health status.

These predictions have not always been matched by the model results. Rarely have most studies even tested for sensitivity of the WTP for different risk changes provided to separate samples (the external scope test), let alone passed them. The recent CV study by Krupnick et al (2000) passes this test but fails the more stringent proportionality test. The same study (as noted above) is the only one to test for the WTP for future risk reductions in a survey also testing the WTP for contemporaneous reductions. This study found the former to be from 15-30% of the latter for average futurity of 19 years, and with an average perceived probability of making it to 70 (when the risk reduction would begin) of about 75%. The empirical studies have all found income effects as expected but have had difficulty separating baseline risk from age, because the two move together. Also, all studies have had very limited participation of older individuals. An exception is Krupnick et al (1999) which had one-third of their sample over 60 years old (up to 75), finding the expected inverted-U relationship, which was steeper than that of Jones-Lee et al (1985); confidence intervals were wide. Finally, only the Krupnick et al study explicitly addresses the effect of health status on WTP. Further improvements are required.

Table 6. **Theoretical predictions and empirical results of studies estimating WTP for mortality risk reductions**

Study	Risk Change	Future Risk Change	Baseline Risk	Income (or proxies)	Age	Health Status
<b>Life cycle model: Theory</b>	+, proportional	-	+ <sup>a</sup>	+	- <sup>b</sup> + then - <sup>c</sup>	indeterminate
<b>Empirical Studies</b>						
<b>Compensating Wage</b>	+	NA	- <sup>d</sup>	+	-	NA
<b>Other Revealed Preference</b>	+	NA	?	+	+	NA
<b>CVM</b>	+, not proportional	-	Varies	+	+ then -; +; -	0 (one study)

Source: Hammitt and Graham (1999) and authors.

- a. Small “dead anyway” effect.
- b. With borrowing against future earnings.
- c. Inverted U with no borrowing.
- d. Self selection by risk tolerant workers.

### 6.2.3 Ratings

Based on the above discussion, as shown in Table 7, none of the approaches for estimating preferences for reducing mortality risks in monetary terms is fully satisfactory (i.e., given an A rating). The latest attempts to use CV approaches designed to fit the “commodity” being valued represent a potentially more credible literature if the results hold up to scrutiny and are replicated.



Table 7. **Credibility ratings for the air concentrations → Mortality valuation pathway**

Approach	Criteria			Rating
	Welfare Theoretic (Y/N)	Numbers of Studies (Many/ Some/Few)	Limitations	
<b>Human Capital</b>	N	M (not recent)	Undervalues non-workers	D
<b>COI</b>	Not usually; in principle could be If separate estimates available for pain and suffering	M	Usually underestimate	C
<b>Revealed preference: Hedonic Labor Market; others</b>	Y	M	Inappropriate commodity/ Population sampled. Many ad hoc approaches to adjust.	B
<b>CVM: non-health</b>	Y	S	Inappropriate commodity/ Population	B
<b>CVM (and conjoint): health</b>	Y (restrictive conditions for conjoint)	F	Needs peer review and replication	B
<b>QALYs</b>	N. Based on medical cost to reduce a QALY	F	At least recognizes that life-years may matter	C

### 6.3 Valuing chronic health

#### 6.3.1 Evaluating estimation approaches

We have identified only two types of approaches that have been used to estimate preferences for reducing chronic morbidity risks and expressing these preferences in monetary terms: the cost of illness approach and stated preference techniques.

The cost of illness approach breaks down the consequences of illness into its component parts and attempts to place values on each part. The ideal WTP measures would capture all the medical costs, pain and suffering, time loss, productivity loss and fear of an illness (see Harrington and Portney (1987) for the basic model). This approach has a welfare theoretic basis but is basically a stop-gap to use when other approaches fail. For instance, Hartunian (1985) uses a model of the progress of cancer and (separately) of respiratory disease to estimate the medical costs from having these diseases over one's lifetime. Cropper and Krupnick (1998) have estimated the consequences of having various chronic diseases on wages and labor force participation. Ultimately, such measures founder on their inability to capture the pain and suffering that are likely to arise from chronic illness.

Conjoint analysis has been used by two studies to value the WTP to reduce risks of developing chronic respiratory disease, and several studies have addressed the WTP to reduce cancer morbidity risks. Viscusi, Magat, and Huber (VMH) (1991) and Krupnick and Cropper (KC) (1992) used conjoint analysis to examine the WTP to reduce the risks of chronic respiratory disease. This analysis involved asking subjects to choose between two cities to live in, where both are preferred to their present city, and the cities differ in the risk of developing chronic bronchitis (or respiratory disease in general) and in one other characteristic: the probability of dying in an automobile accident or the cost of living. An interactive computer program changes the magnitudes of these differences to drive the subject to a point of indifference between the two cities. At this point, the tradeoff between automobile-related death and chronic bronchitis is known, and a statistical case of chronic bronchitis can be monetized by using a value of a statistical life. Alternatively, the tradeoff can be posed as being between chronic bronchitis and the cost of living, allowing the value of a case of chronic bronchitis to be obtained directly. The two studies use the same protocol, except that KC chose a sample of subjects who had relatives with chronic respiratory disease and asked a second set of questions to obtain WTP to reduce risks of a chronic respiratory disease with symptoms *just like their relatives*.

Because the former study (which surveyed 300 people stopped in a NC shopping mall) is thought to capture a more random sample than the latter study, the former is used as the starting point of the VSC calculation in many cost-benefit analyses. However, because this study described a case of chronic bronchitis that was quite severe and the KC study asked people to describe the severity of the cases they were familiar with, the latter study's case severity is thought to be more representative of reality. Thus, the VMH estimate is modified using results from the KC study. The adoption of these studies by EPA for valuation of this endpoint in its key cost-benefit studies has been criticized (OMB memo, September 1999) for basing its choice on two "pilot" studies that are unrepresentative of the population at risk, and for using interesting, but largely unreplicated, approaches for valuation.

Dickie and Gerking (1996) use a household production model to address WTP to avoid skin cancer. This study might be useful to estimate a particular ancillary cost of climate change mitigation policy, i.e., reductions in ambient ozone have been linked hypothetically to increased UV-B radiation, which is linked to skin cancer. In the behavioral model in this paper, people combine consumption goods, goods to reduce harmful effects of sunlight, and time spent in the sun to produce utility. The model is formulated to permit risk perceptions to influence averting behavior, which then influences WTP. In a contingent behavior survey of 300 people eliciting WTP for a lotion that reduced skin-cancer risks, average WTP varied by perceived baseline risk and income, from \$30 to \$50 for a 5% reduction in lifetime skin-cancer risk. At the true risk level (1 in 7), this value is \$44, or \$6,160 ( $\$44/0.007$ ) for a statistical case (in 1988 dollars).

There is also a literature providing QALY estimates for chronic diseases, converting to dollars using some arbitrary cost figure or benchmark, e.g., \$50 000.

### 6.3.2 Ratings

As for mortality risk valuation, the literature has gaps here as well (Table 8), with very few studies even addressing the topic of valuing chronic illness.

Table 8. **Credibility ratings for the air concentrations → Chronic morbidity valuation pathway**

Approach	Criteria			Rating
	Welfare Theoretic (Y/N)	Numbers of Studies (Many/ Some/Few)	Other Limitations	
<b>COI</b>	Not usually; Hospitalization; sometimes labor productivity (which is a revealed preference approach)	M: medical cost studies F: labor productivity studies	Pricing medical services can be difficult where medical care is socialized or subsidized	C-B
<b>Conjoint: health</b>	Y (restrictive conditions for conjoint)	F	Small, non-representative samples	B
<b>CVM: Health</b>	Y	F	Usual issues	B
<b>QALYs</b>	Under very restrictive conditions; but not in practice	S	Can't compare with costs; not sensitive to severity	C

## 6.4 Acute health valuation

### 6.4.1 Evaluating estimation approaches

The ideal WTP measures for WTP to avoid acute health effects<sup>42</sup> would capture all of the medical costs, pain and suffering, time loss, and fear of an acute illness (see Harrington and Portney, 1987, for the basic model). In principle, the stated-preference approach can come closest to reaching the ideal. Three contingent-valuation (CV) studies (Loehman *et al.*, 1979; Tolley *et al.*, 1986a; Dickie *et al.*, 1987) are the original studies of this type. They used bidding procedures to elicit estimated values for respiratory-symptom days, with average estimates ranging from \$5 to \$25, depending on the symptom, its severity, and whether a complex of symptoms is experienced.

All those studies have drawbacks, related mainly to their methodology-the CV studies were performed before many of the most important advances in CV techniques. But they offer consistent ranges of estimates for WTP to avoid a particular type of symptom.

One of the only European studies on the WTP to avoid acute health effects is also a very recent one (Navrud, 1997). Over 1 000 Norwegians were interviewed in-person to ascertain their WTP to avoid a variety of acute health effects (one more day over their usual annual frequency and 14 days over their usual frequency). The values for avoiding symptom effects (in \$1990) are slightly smaller than those found in the older U.S. studies, but the asthmatic values are far larger.

<sup>42</sup> This literature uniformly treats acute health effects in a deterministic rather than a probabilistic context.

A more recent study is that of Alberini, Cropper, and Krupnick (1997), which surveys a sample of 832 Taiwanese about their WTP to avoid their most-recent episode of acute respiratory illness. This approach differs from that of some of the other studies which describe the symptoms and duration of the episode for the person. Statistical techniques are used to relate the Alberini-Cropper-Krupnick values to the duration and severity of the episode and other variables. The authors find that avoiding a one-day episode of a cold like they had recently experienced is valued at about \$20 (1992\$) with the marginal value of avoiding longer episodes falling rapidly to \$6 at 5-days. For non-cold episodes, the value for a day was \$31. The results also are compared with those of the older studies. Even though the Taiwanese have lower incomes than the US people participating in the US studies, the Taiwanese WTP bracketed the Taiwan WTP estimate without any adjustment for per capita income differentials. This not only raises a question about benefit transfers adjusted for income differentials but raises another problem: when there is more than one study from the source country to use in a benefits transfer, and the studies have different results, how does one choose the studies to make the transfer?

There are several non-CVM alternative approaches to valuing acute health effects. One—the cost-of-illness approach—attempts to tally the various out-of-pocket costs associated with illness. By missing “pain and suffering,” this approach necessarily underestimates costs (benefits). Hospitalization, emergency room, doctor, and drug costs (including charges paid by insurance companies), the value of nonwork time spent in these activities and being sick, and the value of work lost as a result of illness are the categories of costs most often estimated.

Another, less-used approach—the averting-behavior method—attempts to infer the WTP to avoid a health effect by observing and placing a value on behavior used to avoid the health effect. For instance, if someone stays indoors with the air conditioner on all day *because* of high pollution, the added costs of the electricity bill might be related to the WTP to avoid the health effect. For this approach to yield defensible estimates of value requires a number of stringent assumptions, e.g., the air conditioner has no other attributes than cleaning the air. In practice, it is little used, particularly in an acute-health context.

#### 6.4.2 Ratings

Here again, the valuation studies leave much to be desired (Table 9)—a particularly distressing state of affairs, given how relatively easy a study of this type of endpoint would be to mount. Work on these endpoints in Europe may revitalize the field. However, unless health scientists find far larger effects of pollution on acute health than found so far, the value gained from many more valuation studies may be quite limited.

Table 9. Credibility ratings for the air concentrations → Acute morbidity valuation pathway

Approach	Criteria			Rating
	Welfare Theoretic (Y/N)	Numbers of Studies (Many/ Some/Few)	Other Limitations	
COI	No	M	Pricing medical services can be difficult; May not need any	C
Revealed preference (averting behavior)	Y (under restrictive conditions)	F		B
Conjoint: health	Y (restrictive conditions for conjoint)	F		B
CVM: health (old)	Y	S	Old methods/studies; some ad hoc estimates; small samples	B
CVM (new)	Y	F	?	?
QALYs	In theory, under very restrictive conditions; rarely in practice	M	Scores not very sensitive to severity	C

## 7. Developing country issues

### 7.1 Valuation

Probably the most controversial aspect of estimating ancillary benefits is the link between health effects and valuation in developing countries. Indeed, previous IPCC assessments have sparked great controversy, because they have presented non-market values for health improvements in developing countries that, under some assumptions, appeared to value lives of populations in such countries lower than those in developed countries.

Many of the criticisms of such valuations are criticizing valuation *itself*, as much as its application to developing countries. Nevertheless, health improvements can be thought of as commodities in the broadest sense. That is, we buy health improvements (or risk reductions) in our everyday life. We buy medicines and medical procedures, we spend time exercising, preparing low calorie meals, etc. As we do for commodities, we trade-off income or time against health commodities. We speed on the highway to save time, when we know it will cost us a bit in gas and raise our risk of injury, property damage and death. Thus, thinking of health improvements and risk reductions as a commodity, it is easy to imagine that people have a willingness to pay for this commodity, i.e., it has price sensitivity, where WTP varies depending on how large the risk reduction is.

Let's consider prices more generally. Prices of given commodities vary across countries and, of course, even regions within a country. Even very homogeneous, internationally traded commodities, such as oil, vary in price across countries—in part because of transportation cost differentials. Price differentials also arise when pure competition is not present (either at a country or firm level). But in the long run, even with pure competition on locally traded goods, price differentials arise from the interplay of supply and demand. And underlying demands are preferences—which depend on a host of individual, family, cultural and other factors, as well as income. Thus, there is no reason to expect the demand schedule for one country for the commodity of “risk reduction” to be equivalent to those of other countries.

Some feel that using income as a means of adjusting demand for health improvements would disadvantage the low income countries. But, their level of health protection depends on the cost as well as the preferences for health protection. There is no reason to believe that health preferences fall proportionally to income.

Thus, our view is that there is not a qualitative difference between valuing health in developed and in developing countries. In both cases, people must make—and do make—tradeoffs in their daily lives between obtaining reductions in health risks and having more of something else, be it money, the goods that money can buy, their time, or something else of value to them. The job of the ancillary benefits analyst is to recover the preferences underlying those tradeoffs and give them standing in the analysis—whether in the form of monetary values, or some other values.

This is not to say that valuation of health improvements in developing countries is no more challenging than doing so in developed countries. Developing countries in general exhibit a number of specific characteristics relative to developed countries that exacerbate the estimation of ancillary costs/benefits. Indigenous valuation studies are in very short supply, basic data are limited, exchange processes are constrained, growth is (hopefully) rapid, and a number of broader social development concerns are more vexing and severe, such as living conditions of the poor and institutional capacity needs.

For estimating preferences for health improvements, these characteristics have many implications. Among them (see Markandya, 1999):

- Reliance on valuation studies performed in developed countries (or the unit values arising from such studies) will introduce more than the usual amount of uncertainty to valuation estimates
- Limited basic data, such as data on wage rates in manufacturing, make application of some valuation approaches problematic
- Medical cost information may not reflect social opportunity costs and, therefore, may need adjustments to make them more reliable
- Hedonic labor market studies, which presume that labor and goods markets are competitive and workers have reasonable information on death and injury risks, may carry more uncertainties than those of developed country labor markets
- Valuation of health of household members —particularly children—may be quite different than in developed countries because of the more central role played by children in the economy and the household

- Rapid economic growth means preferences are changing as well, raising the questions about the applicability of indigenous studies several years hence. Perhaps, benefits transfers become more legitimate over time.

It used to be the case that valuing health improvements in developing countries would require performing a benefits transfer—using concentration-response and valuation information from a developed country (usually the U.S.) where such studies had been mounted and applying them, sometimes with adjustments, sometimes not, to the developing country. This practice is the one that led directly to the controversy noted above. However, the process is much more broadly applied, being used to “transfer” benefit estimates from one part of a country to another and from one developed country to another developed country, as well as to a developing country.

Moreover, as we show in Annex I, environmental economists and others have begun to perform primary valuation research in developing countries, which has revealed that preferences for health improvements, conditioned on income as they must be, are not terribly different between developed and developing countries. Furthermore, comparisons between values obtained through benefits transfer and values obtained directly from the people in the country reveal that such transfers are not particularly reliable. Thus, benefit transfer techniques may be gradually supplanted by direct analyses.

The above phrase “conditioned on income as they must be” highlights one of the more controversial, basic elements of modern welfare economics—that equity considerations should be kept separate from efficiency. See Krupnick, Burtraw and Markandya (2000) for a discussion.

## 7.2 *Transfer Uncertainty*

The idea behind benefit transfers is to fill in gaps in the availability of information on, in this case, the preferences of individuals in a country for health improvements. The underlying theory is in Desvousges et al (1985, WRR), where a hierarchy of approaches is suggested. They begin with simply using, say, a value of statistical life estimate in the source country and applying it to a receiving country’s population. Simple types of adjustments, for example, income differentials, characterize the next approach, moving to more complicated adjustments that incorporate a host of differences between the source and receiving country, such as education levels, baseline risks, age distributions, health status, and the like. The income adjustment is usually made by using an income elasticity of willingness to pay for the improvement. Using an elasticity of 1.0 would change the WTP in the receiving country proportionally to the relative per capita income differentials across the two countries. An elasticity of zero would mean that no adjustment was made for income differentials.

A number of health valuation studies have used income differentials within their sample to calculate income elasticities of WTP. Surprisingly, these studies find relatively small elasticities, ranging from 0.2 to 0.6 (Mitchell and Carson, 1986; Alberini, et al, 1997; Loehman *et al.*, 1979).

Using an elasticity of 0.3, for example, to adjust WTP across countries would imply that the WTP of a receiving country with a tenth of the per capita income (in purchasing power parity terms) of the source country would be 73 per cent of that of the source country’s WTP. For the example above, the \$100 WTP in the U.S. is translated into a \$73 WTP in Ukraine when the elasticity is 0.3. The appropriate translation of the willingness-to-pay (WTP) from the study country (S) to the target country (T) is given by:

$$WTP_T = WTP_S (1 - \epsilon_{WTP,Y} (Y_S - Y_T) / Y_S)$$

Where  $\epsilon$  is the income elasticity of WTP with respect to per capita income, and  $Y_T$  is the income of people in the target country.

There are a number of concerns about using this approach. First, it ignores many other factors that might make WTP different across countries—cultural factors as much as more measurable variables, such as age distribution. Second, it uses individual variation in incomes within what is usually a basically wealthy source country to estimate WTP for a country with low average incomes. This is out of sample and ignores differences in income distribution.

In principle, the more complex adjustment approach can be parameterized by using a single study, estimating the variation in WTP as a function of a number of appropriate regressors, such as education, etc., although for many such studies, microdata of this nature is not generally available. The CV studies generally can provide such estimates.

Because of the lack of microdata for many of the original valuation studies, some benefit transfer studies have used the valuation studies in a meta-analysis. By treating each study's results as an observation, group data (say on average education levels in the country where the study was performed) can be used to estimate the effect of variables on WTP. Bowland and Beghin (1998) for instance, use 33 labor market studies to estimate the marginal effect of risk differences, income, education, age, and various labor market descriptors on WTP for mortality risk reductions and apply the estimated equation to estimating benefits of fine particulate reductions in Santiago. They find that the implied VSL is about \$740 000 U.S. in \$1992, adjusted for PPP, compared to the U.S. standard VSL figure of \$4.8 million (\$1990).

This study has some problems—many of the observations use the same labor market/mortality dataset, key variables to help explain WTP differences (such as health status) are missing, and the studies are mostly for the U.S., so those have insufficient variation in many of the variables. More importantly, one of the key results—that the income elasticity of WTP is well over 1.0—is much at odds with the rest of the literature. Chilean income is assumed (in this 2010 analysis) to be about half of U.S. income. But the high income elasticity (plus other differences between the countries) serves to widen the gap to a ratio of 1 to 7.

Another literature “kills two birds with one stone,” by estimating values directly from an indigenous study and then using benefit transfer techniques to test for whether these techniques can come close to replicating the values from the indigenous study. One example is a comparison of a series of wage compensation studies done in India and Taiwan compared to transferred values from U.S. studies (Simon, Cropper, Alberini, and Arora, 1999) (Table 10).



The authors cite Shanmugan (1977) as the first study to examine the Indian manufacturing industry, although focused on one metropolitan area and restricted to blue collar males working in industries described at the “two-digit” level. He estimates a VSL of \$400 000 on an exchange rate basis. Simon, Cropper, Alberini, and Arora (1999) take a broader, more comprehensive look at Indian labor markets. They examine blue-collar workers, both male and female, in all of India’s manufacturing industry and at the three-digit level, using risk data at the industry, rather than the occupational level. They find that the implied VSL is between \$150 000 and \$360 000 on an exchange rate basis (\$1990). They compare the VSLs across countries by computing the ratio of the VSL to the present value of foregone earnings implied by a unit change in occupational death risk—i.e., to the values derived using the human capital approach. They find that, using their original study, the VSL is from 20 to 48 times larger than the value of foregone earnings. The implication is that a benefit transfer assuming a unitary income elasticity of WTP would underestimate the VSL for India by 50-60%.

They also cite Liu, Hammitt, and Liu, who used five years of wage and accident data from nonagricultural sectors and the three-digit industry level, using risk measures by industry. They find VSLs that range from \$135 000 to \$589 000 (\$1990) depending on the year of the data, and average about \$410 000. These authors also present results of a meta-analysis using the VSL estimates reported by Viscusi (1993) with per capita income and annual fatality risk as arguments. Using this log linear equation, which features an income elasticity of about 0.5 (insignificant) and a risk elasticity of -0.27 (which is significant) yields a VSL of \$1.4 million. The implication is that a benefit transfer using a unitary income elasticity of WTP and accounting for baseline mortality risk differences would overestimate a VSL for Taiwan by 3-4 times.

**Table 10. Wage compensation studies in U.S., India, and Taiwan and benefit transfer comparisons**

<b>Country/Transfer</b>	<b>VSL (\$)</b>	<b>GDP per Capita (1995 US\$)*</b>	<b>VSL/GDP Per Capita</b>	<b>VSL/PDV foregone income</b>
USA	4,800 000	26,037	184.35	8-23
India (Shanmugan, 1977)	400 000	365	1096	73
India (Simon, et al, 1999)	150 000- 360 000	365	411-986	20-48
Taiwan	410 000	11,276	36.4	7-8
USA to Taiwan Function Transfer (Viscusi, 1993)	1,400 000	11,276	124.2	24-27

\* - Statistics for USA and India from the United Nations “World Statistics Pocketbook, 1995.” Statistics for Taiwan from Directorate General of Budget, Accounting, and Statistics, Republic of China.

Another example is Chestnut, Ostro and Vithit-Vadakan (1997), who administered a contingent valuation survey to 141 adults (half nurses) in Bangkok to estimate their willingness to pay for avoiding one respiratory-symptom day of three alternative types (no activity restriction, partial activity restriction, and work loss day). Using an exchange rate conversion (25 baht/\$), mean estimates were \$16, \$30, and \$63 for no restrictions, partial restrictions and work loss respectively, while the medians are \$4, \$12, and \$26, respectively. Standard median estimates used in U.S. studies (Table 6 above) for a symptom-day with no restrictions range from \$12 to \$15, while restricted activity days range from \$30 to \$50. With U.S. median incomes from 3 to 4 times larger than those of respondents in the Bangkok sample, it is not clear that benefits transfer based on unitary income elasticities would be a poor proxy for actual WTP.

On balance, though, the main conclusion of these comparisons is that benefit transfers assuming the income elasticity of WTP is 1.0 or even making other adjustments do not appear to be reliable for valuing mortality and morbidity risks in developing countries.

## **8. Conclusions**

The IPCC is charged with assessing current technical knowledge about likely physical, social and economic changes that will occur as a result of climate modification over the next century. The matter of how various climate policies will affect public health has remained beyond the purview of climate models and has received little attention in the Special Reports on Emission Scenarios. In addition, this aspect of climate change policies has yet to rise to prominence in international climate mitigation policy negotiations. Although some have suggested that the size of potential ancillary benefits could be as great as some of the costs associated with climate change mitigation policies, others find relatively small effects relative to costs (Grubb, 1999).

Estimating the ancillary public health consequences of greenhouse gas mitigation policies is a challenging, multidisciplinary task, drawing upon expertise in economics, emission inventories, air pollution modeling, and public health. This task is made easier by recognizing that such estimates are chiefly conducted in order to compare to the costs of various proposed greenhouse gas mitigation strategies. Both ancillary benefits and mitigation costs are contemporaneous and may, in the first instance, be analyzed at a country-level.

Based on available information to date, the estimated improvements in public health linked with various reductions in air pollution reductions appear to be significant-both in developed and other countries. However, these estimates are uncertain and do not reflect the full array of pollutants that are likely to be involved, nor do they address myriad other pathways potentially affecting public health beyond those tied with air pollution. Notwithstanding these uncertainties, the analyses of ancillary health benefits can provide useful information to the policy debate about the scope, design, and timing of climate policy.

We want to emphasize that it is one thing to link changes in air pollution and health and another thing to link greenhouse gas mitigation policies with specific changes in air pollution. Our paper has focused only on the former.

In summary, we offer the following major points:

1. While the public health benefits of GWG co-pollutant reductions are ancillary to the direct weather effects of GWG's, they are more immediate in time than the climate changes (i.e., mostly begin as soon as emissions are reduced), and therefore they gain increased importance when effects are viewed on a Net Present Value basis.
2. In terms of monetized ancillary benefits, the air-mortality pathway is the dominant pathway in analyses to date. An annual unit ( $\text{ug}/\text{m}^3$ ) reduction in  $\text{PM}_{2.5}$ , for instance, can generate benefits of about \$250 per person based on the Pope study and the standard VSL. Further, there is relatively low *statistical* uncertainty around this estimate. However, the dominance of air pollution-mortality monetary valuations may not remain once a broader assessment can be conducted.
3. There is a need to recognize the broad array and large number of adverse health effects that are associated with air pollution, many of which are not yet quantifiable from either a health effects or valuation perspective. Also, it should be noted that, if non-monetary valuation metrics, such as numbers of effects or quality of life changes are considered, less severe outcomes will dominate the costs/benefits.
4. PM is the air pollutant that most recent studies indicate has the greatest public health consequences. But PM composition and size is heterogeneous across sources. There is some evidence that PM from different emission source categories may have different health implications. GHG mitigation strategies that lead to lower emissions of fine particulates, and that result in reductions in coal combustion, industrial processes, or diesel fuel combustion may lead to greater ancillary benefits than those that reduce larger particles or are achieved via reductions in low PM sources, such as natural gas combustion.
5. However, there remain uncertainties with respect to the PM-mortality relationship. Of these, probably the greatest uncertainty is associated with the question of the possible contribution of co-pollutants to the PM effect estimates (i.e., to what extent PM is serving as an index of air pollution, in general), and the potency of specific PM components.
6. The air-chronic respiratory disease pathway is an additional important pathway. There is an emerging epidemiological literature on this endpoint, with substantial model uncertainty. Model and statistical uncertainties are especially large in the valuation literature regarding these endpoints.
7. The standard VSL is drawn from labor market studies, which may be inappropriate for this pathway. Contingent valuation approaches have the potential to yield appropriate estimates. However, these methods have had difficulty meeting the theoretical expectation that WTP should increase proportionally with greater risk reductions. Even the best studies suffer from this problem. Valuation of children's health and estimation of the WTP for improvements in other people's health are active areas of research.
8. Although uncertainties in valuation are large, there is fairly broad consensus over the values to be used in air pollution-health benefits analyses, as evidenced in Table 5.

9. No matter what approach is taken to estimate the public health consequences of air pollution reductions, benefits are likely to be substantial. In order to conduct the most robust and uncontroversial analyses, we recommend relying on the most defensible, transparent methods, even if they are recognized as being deficient to the conceptually correct methods. In other words, a strategy of retreating to defensible borders seems appropriate. For example, such measures may indeed be the standard VSLs from labor market studies, although they probably overestimate benefits/costs. Still better would be to make some adjustments for age and health status or wait for the new CV literature to develop further. In some cases, the human capital approach might ultimately be the most defensible. These approaches may all be superior to benefit transfers.

Overall, though still a work in progress, the present techniques available for the analyses of the ancillary public health costs and benefits are adequate and appropriate for implementation by those comparing the relative merits of various GHG mitigation policies. The public health changes associated with GHG mitigation strategies should be considered as a key factor in the choice of GHG mitigation strategies made by the nations of the world in the next few years.

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## ANNEX I

### SOME RELEVANT EPIDEMIOLOGIC AND ECONOMIC BASICS

#### **Epidemiological concepts**

Epidemiology is the study of the distribution of determinants of disease frequency in human populations over time and in specific areas. Epidemiological studies statistically evaluate changes in the occurrence of adverse health effects in a single population as it undergoes varying real-life exposures to pollution over time, or across multiple populations experiencing different exposures from one place to another. There are two types of epidemiologic studies: 1) experimental studies, which involve the deliberate application or withholding of a supposed cause and observation for the subsequent appearance or lack of appearance of effect (eg., intervention studies), and; 2) observational (non-experimental) studies, in which nature determines the exposure in real-world situations. In most cases, past analyses of ancillary benefits of energy policy or regulatory actions have relied primarily on available observational epidemiological studies to derive quantitative relationships between changes in environmental conditions and changes in public health.

Within observational studies, two major types of effects have been looked at in assessment of environment-health effects relationships: long-term (chronic) exposure studies, and: short-term (acute) exposure studies. The former compare individuals or populations over long periods of time so that they provide information about the cumulative effects of pollution, but they are challenging to conduct, and can be confounded by differences in other characteristics (e.g., socio-economic conditions) across individuals or populations. Individual-level studies are preferable to population-based cross-sectional studies, because it is easier to control for potential confounders on an individual level. Most published studies of the effects of air pollution, however, have looked at acute effects using time-series methods, which follow the exposure and health effects of a population over time. Time-series studies have the major advantage that many potentially confounding factors in cross-sectional studies, such as socio-economic status, are relatively constant over time, and the population under study serves as its own “control”. This obviates the need to separately analyze a comparison population, and all their respective associated (and potentially confounding) differences in characteristics. However, time-series models have their own challenges, foremost of which is that they are often complicated by the fact that other environmental factors, such as seasonal variations, may confound the results, if not addressed in the statistical analysis. Thus, epidemiological analyses are complex, but can provide among the most useful and relevant information available regarding the relationship between environment and health.

Most of the recent studies linking air pollution and health have applied multivariate methods, such as Poisson models, that statistically address major potential confounders. In such models, the mean of the daily effects ( $Y$ ) is modeled as an exponential function of the explanatory variables ( $X$ ):

$$E(Y) = \exp[\beta * X]$$

The relative risk (RR) associated with a change in the air pollutant concentrations (one of the  $X$  variables) is given by:

$$RR(\Delta P) = \exp[\beta * \Delta P]$$

The slope coefficient,  $\beta$ , is a measure of the health effect per unit pollution, and  $\Delta P$  is the change in air pollutant concentrations from a reference concentration,  $C_0$  (e.g., a no GHG mitigation case projection). The relative risk shown above can be applied to the base rate of effects ( $R_0$ ) in a population of interest that is being exposed to air pollution (e.g., number of deaths per 100 000 persons during a baseline year) in order to estimate the projected effect of a pollution concentration change. Therefore, as discussed in Cifuentes *et al.* (1999), in the case where no threshold of effects applies, for an increase in concentrations  $\Delta P$  from  $C_0$  to  $C$ , the change in health effects ( $E$ ) (e.g., in deaths per year) is given by:

$$[(\Delta E)] = [\exp(\beta * (C - C_0)) - 1] R_0 \cdot \text{Pop}$$

where  $R_0$  refers to the number of effects at concentration  $C_0$ , and is generally obtained from available public health statistics data. For short-term effects analyses, like the daily time series studies, the above formula applies to daily effects, and the health effects rate should be expressed as the number of effects per day. To obtain the number of excess effects in a year, it is necessary to add the effects for all days of the year, assuming that short-term “harvesting” of deaths is not a major component of the acute air pollution effect (Schwartz, 2000).

## Economic concepts

### Definitions

**Ancillary benefits and costs** are changes in social welfare that arise as a result of (1) externalities and (2) government and other market failures when undertaking GHG mitigation programs. **Social welfare** is, in the abstract, the summation of all the things that members of a society see as contributing to the quality of their lives, individually and collectively—without enumerating what those factors might be. Social welfare is meant to be a yardstick that permits us to look at our society in alternative states of the world and choose the state in which we are best off. Positive changes in social welfare are called benefits; negative changes are called social costs.

The **social cost** of producing a good or service is given by the **opportunity cost** of all the resources that go into producing it. Some of these may not involve financial payments. Hence the **financial cost** may not be equal to the social cost. The financial cost is equal to the private cost if all resources provided by the party responsible for the good or service are paid for in money.

For instance, the financial cost of supplying electricity generated from a coal-fired power station will include payments to labour, capital and raw materials. This will not equal the social cost, however, if (a) the payments are not based on the opportunity costs of the labour, capital etc., say because the power plant operates in a country where labour markets are not competitive; and (b) resources such as clean air and water have been used up in the production of the electricity and payment is not made to those affected by the loss of those resources, based on its opportunity cost. The financial cost can also be referred to as the **private cost** of supplying the electricity if all resources under the control of the supplier are paid for in financial terms. If some resources (e.g. own labour) are not so paid for, the financial cost may differ from the private cost.

One of the most important reasons why the financial or private cost may differ from the social cost is the presence of **external effects or externalities**. Externalities are said to arise when the production or consumption of something has an impact on welfare which has not been fully taken into account by those responsible for production or consumption decisions. In the above example, the welfare costs of air and water pollution from the generation of the electricity may not be taken into account by the suppliers of electricity. To account fully for this welfare effect, persons affected by the loss of air and water quality would have to agree to the losses based on their WTA payment. This presumes that they are fully informed about the nature of these losses and that this information is well established and publicly accessible.

The discussion so far has been about externalities, or situations where missing markets cause a divergence between private and social costs. Such divergences can also arise, however, for other reasons. Prominent among them are:

- Government subsidies and taxes.
- Government controls that restrict, in one form or another, the supply for demand for a particular input or output.
- Market imperfections such as monopoly or monopsony power.

All these factors result in market prices deviating from marginal social costs. Hence in making a proper assessment of the social cost this divergence has to be allowed for. The following are some important examples of divergences and how they may be addressed.

Adjustments for deviations between private costs and opportunity costs in labour and capital markets are made through the use of '**shadow prices**'. As an example, if the wage paid to a worker is \$30/day and the opportunity cost of his or her time is only \$15 a day, a shadow price of 0.5 is applied to the actual wage to get the social cost of that input. The underlying imperfection in this case may be union power which keeps the wage artificially high, or macroeconomic failure, which prevents the labour market from clearing. Likewise, where capital markets are distorted, the market price of capital may not reflect its true scarcity. This would imply the need to apply a shadow price to capital of greater than one; something which is routinely done in project appraisal by public sector investors. The details of how such premia can be calculated are discussed in standard treatments of the subject (see, for example, Ray (1984)).

The standard assumption for social cost pricing of all goods and services is to take the international prices<sup>43</sup> for all tradable commodities. This assumes that international prices are indeed free market prices. If that is not the case, an adjustment must be made to the international price to reflect any divergence from social cost. Note that such adjustments will mean that all inputs and outputs will be valued net of any taxes or subsidies.

### *Alternative to Money Metric: QALYs*

A measure is needed for expressing preferences for health improvements. Otherwise, we have no way of judging whether one state of nature is more desirable than another state of nature. Owing to its convenience and familiarity, money is the standard numeraire for expressing preferences for health improvements and aggregating over different health improvements. But it needn't be the only one. Recently, the Quality Adjusted Life year (QALY) and a close relative, the Disability Adjusted Life year (DALY) have gained popularity as a way of expressing preferences and adding up over alternative health effects.<sup>44</sup>

The Quality Adjusted Life years (QALY) approach assigns numeric values to various health states, so that morbidity effects can be combined with mortality effects to develop an aggregated measure of health burdens. Although there are variations to the QALY approach, there is agreement on basic elements. Health states are measured on 0-1 scale, usually with 1 representing a year of completely healthy life and 0 representing death. A year of suffering some specified illness is weighted somewhere between 0 and 1. For example, a year of extreme pain may be valued at 0.5. A basic assumption is that the QALY values are additive, so that a treatment that eliminates extreme pain for one year for two individuals ( $2 \times 0.5$ ), is equivalent to a treatment that adds one healthy year of life. Life years are treated equally for all individuals, so that a single healthy year is weighted the same regardless of age or income (this is known as the "egalitarian assumption"). Weights can be based on the preferences of individuals (through rating scales, standard gamble, and time trade-off approaches), but a recent survey of QALY studies found that this has often not been the case; in many studies, physician judgments have substituted for individual preferences (Neuman et al, 1997). Both the standard gamble and time trade-off approaches are consistent with utility theory as long as some fairly restrictive assumptions are met.<sup>45,46</sup>

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<sup>43</sup> The relevant prices are the international price f.o.b. from the country concerned for goods that are exports or potential exports, and c.i.f. to the country for goods that are imports or import substitutes. Where goods are not tradable, shadow prices are estimated using the costs of production with inputs priced at international prices.

<sup>44</sup> The National Institutes of Health have examined whether their allocation of research dollars is correlated with DALYs from different diseases and conditions. [ ] QALYs have been used as the basis for an EPA sponsored study of the risk-risk trade-offs of improving drinking water quality. The Global Burden of Disease is a study by the Harvard School of Public Health that attempted a comprehensive assessment of mortality and disability from diseases, injuries and risk factors in 1990 and projected to 2020 using DALYs (Murray and Lopez, 1996).

<sup>45</sup> Constant proportional trade-off (if an individual is indifferent between 30 years of perfect health and 40 years of illness, they are also indifferent between 15 years perfect health and 20 years of illness) and utility independence (the ratio of utilities in two health states is independent of the time frame chosen).

<sup>46</sup> A variant of the QALY, the disability-adjusted life-year (DALY) has been developed by researchers at Harvard University to estimate the global burden of disease (cite). There are two important differences from the QALY.



### *The Damage Function Approach*

The standard approach to estimating health benefits of changes in pollutants is the Damage Function Approach (DFA). In the health context, this approach involves deriving a mathematical relationship between pollution concentrations that affect health,<sup>47</sup> and the various types of health effects and between the health effect and individual (or social) preferences for reducing the risk or the incidence of this effect. Thus, the estimation of health benefits of changes in pollution requires an understanding and careful integration of health science with economics. This integration involves matching as closely as possible the starting point of the valuation analysis to the end point provided by health science—a health response (such as a symptom-day or an increase in mortality risk) or a health consequence (such as a hospitalization or a bed-disability day). In addition, it requires knowledge of the population by cohorts that map into the health endpoints (e.g., asthmatics) and information about these populations (such as mortality rates).

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<sup>47</sup> The step from emissions to concentrations precedes this step and is not unique to health. However, to go beyond this step requires that the pollution concentrations be of the appropriate type and time measure (average of the daily one-hour peaks, annual average, etc.) to link to the concentration-response functions.

## ANNEX II

### VALUATION STUDIES IN DEVELOPING COUNTRIES

*Labor Market Estimates of WTP for Mortality Risk Reduction.* (see text above)

*WTP for reduced respiratory symptom-days.* See Chestnut, Ostro and Vithit-Vadakan (1997) and Alberini *et al.* above.

*WTP and COI for vaccinations against malaria.* Cropper *et al* (1999) estimate the willingness to pay to protect against malaria for a year using a contingent valuation approach and comparing this to cost of illness measures. The COI measures include treatment and drug costs, transportation costs, work days lost and losses associated with household labor substitution to arrive at a cost of from \$4-\$24 (1997\$) per episode, \$9-\$31 per household annually. In contrast, annual household WTP for preventing malaria \$36 (median of \$25), which is about 15 per cent of annual income. Under a range of assumptions, WTP is from 2-3 times larger than COI. The income elasticity of vaccine demand is 0.4. The authors also estimated the WTP for bednets, which provide partial protection from malaria. They find the WTP is about 70 per cent of the WTP for a vaccine promising complete protection. The comparison may be a bit misleading however, because the bednets would normally provide protection over several years.

*WTP for Medical Care.* Gertler and Hammer report estimates of price elasticities for medical services from 10 developing countries and 12 separate studies. These elasticities vary for low income and high income households, the former being about twice the latter, suggesting that, first, the poor's demand for medical care is sensitive to price and second, that the rich are less sensitive to price than the poor. A number of these studies used travel costs to measure price variation as medical services to the poor are often offered for free. This literature (for instance, Lavy and Quigley, 1991; Alderman and Lavy, 1996) also finds that the poor are willing to pay for more accessible and reliable health care and drugs, although are willing to pay less than the rich for equivalent health care.

*WTP for Improved drinking water quality.* Diarrheal disease is one of the major causes of childhood mortality in developing countries. Therefore, observing the steps parents take to improve the quality of their drinking water provides some indication of the WTP to reduce child mortality risks (of course, it also embeds the WTP to reduce the morbidity associated with diarrheal disease. In the U.S., there are a variety of studies examining the WTP for drinking water improvements. Household WTP to reduce organic chemicals in groundwater, for instance, which could reveal a WTP to reduce cancer risks, is over \$1 000, gauged by their averting behavior expenditures. A very different measure of averting behavior comes from Harrington, et al who find that costs to treat water contaminated by giardia range from \$1.13 to \$3.60 per person per day. Households in Korea were found to be willing to pay \$3.28 per month for a device on their public water supply to provide early warning of chemical contamination. McConnell and Rosado (1999) find for a sample of Brazilian households that willingness to pay for filters to make the water safer (a lower bound measure) is about \$10 per month per household.

*WTP for air pollution reductions.* Whittington and Wang surveyed 514 people in Sofia, Bulgaria to assess their WTP for reductions in air pollution (“most air pollutants would be reduced by approximately 75%). They were told that respiratory illness would be reduced (no specified amount), 700 lives per year would be saved out of 1000 who die from health problems related to air pollution), and a variety of other improvements would occur, such as reduced materials damage, improved visibility, and lower damage to trees and plants, and that these benefits would be realized “in a few years time.” Mean WTP was estimated in several ways, the highest WTP being about 4.2 per cent of income. Income elasticity of WTP was about 0.27, in line with other studies. The complexity of the “commodity” being valued and the large benefits being hypothesized makes these estimates not comparable to those of studies that focus on the health effects themselves and use much smaller risk reductions. The bid elicitation method, which asks for the probabilities that the individual would vote yes given a large number of alternative monthly payments, also makes this study difficult to compare with others.

*Demand for Water.* Observing the demand for water in rural areas of developing countries is an indication that even the poorest people have a willingness to pay for important commodities. A large number of studies estimate the demand for water in poor rural communities in developing countries and find that people are willing to pay for better water. North and Griffin (WRR, 1993) used data from rural households in the Philippines to estimate a hedonic property value model for the willingness to pay a private versus a public water source. Households were divided into low, middle and high income households. Households in all income groups were willing to pay “about half their monthly imputed rent for piped water in their house.” This finding suggests that income may not be the best surrogate measure for benefits transfers. WTP is actually largest for middle income households: \$2.25 per month (1978\$ at 7.4 peso/\$) versus \$1.95 for high income households. Quigley (1992) found that low income households in El Salvador were willing to pay 10% of their rent for piped water. Whittington et al (1990) found benefits to rural Kenyans of piped water of 2.5% of their income, based on savings in time from not hauling water.

### *Human Capital/Cost of illness measures of death*

PWGM report on studies using cost of illness measures. Hospitalization costs in Brazil were estimated to be \$5600 per death. Foregone output costs divided them by number of deaths yielded a “VSL” of \$5 000 to \$25 000 (p. 98).

Ortuzar, Cifuentes, and Williams (1999) note that human capital estimates in Chile were recently updated from \$42 000 (1989\$) to \$60,900 because of income increases.

There is probably little need for extending analyses of health benefits/costs outside of the air-mortality pathway. Based on the current literature, best estimates are clearly large. An annual unit ( $\text{ug}/\text{m}^3$ ) reduction in  $\text{PM}_{2.5}$ , for instance, can generate benefits of about \$250 per person based on the Pope study and the standard VSL. Further, there is little statistical uncertainty around this estimate.

However, there are major model uncertainties with respect to the size and composition of PM constituents that are affecting health and how other pollutants should be treated to avoid double-counting. Some strategies for reducing carbon are likely to reduce all conventional pollutants in roughly proportional amounts, making this uncertainty moot. Other strategies could result in changing the proportions of different types of pollutants or, with the class of particulates, the sizes of types, having unexpectedly larger or smaller effects on health.

# ESTIMATING ANCILLARY IMPACTS, BENEFITS AND COSTS ON ECOSYSTEMS FROM PROPOSED GHG MITIGATION POLICIES

by Dale S. ROTHMAN

## Preface

The primary question to be addressed in this paper is what are the ancillary impacts, both positive and negative, on ecosystems from proposed GHG mitigation policies. Several secondary questions follow from this, including what are the best ways to estimate these impacts, can we attach values to these impacts, and how might the information about these impacts best be used to assist policy-making.

It is important to preface what follows with a brief discussion of several caveats on what I propose to do in this paper. Adopting an approach similar to that in the EPA's retrospective (US Environmental Protection Agency 1997) and prospective (US Environmental Protection Agency 1999) studies of the Clean Air Act and its amendments, I will address *changes* in impacts on ecosystems from *changes* in policies. This is a "marginal" approach, in that I do not attempt to draw conclusions about the total value of ecosystems or the total impact that human actions may be having on them.

Secondly, in the context of valuation, this paper is inherently anthropocentric. As noted by Pearce (1992, p.7), "What {economic} valuation does is to measure *human preferences* for or against changes in the state of the environments. It does not 'value the environment'. Indeed, it is not clear exactly what 'valuing the environment' would mean" (author's italics, my brackets). This is not to argue that other forms of value independent of humans, e.g. intrinsic value, do not exist. However, I believe, by their very definition, that neither I nor any other person is privy to truly understanding and putting words to these values.<sup>48</sup>

Thirdly, I will focus, at times, on economic and specifically monetary valuation, but not to the exclusion of other concepts of value and valuation. Policy-making is and should be heavily and appropriately influenced by economic considerations, but not to the exclusion of other concerns (Vatn and Bromley 1995; Toman 1999). Too often, however, an overemphasis on monetary valuations has placed a straitjacket on analysts and limited the information used, at least implicitly, in making policy (Porter 1995).

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Toman (1999) discusses this same issue in somewhat more depth in laying out a general value typology, where he describes four separate forms of value: anthropocentric instrumental, anthropocentric intrinsic, non-anthropocentric instrumental, and non-anthropocentric intrinsic.

Guided by these principles, I hope to contribute positively to the evaluation of proposed GHG mitigation policies, while avoiding certain pitfalls that lurk in linking economic and ecological analysis. By limiting myself to, primarily, marginal changes, I remain within the purview in which economic analysis is most appropriate (Toman 1998; Heal 2000). By not being bound by the specific requirements of monetary valuation, I do not limit myself to a small number of impacts to the exclusion of the vast majority (Rothman 2000).<sup>49</sup>

## **Abstract**

*Many of the policies being proposed to address the issues of greenhouse gas emissions and atmospheric concentrations strongly resemble, if not duplicate, policies that have been proposed and/or implemented to address other environmental issues. At the same time, other policies involve direct changes to ecosystems. Thus, it is reasonable to expect that GHG mitigation strategies will have direct and indirect impacts on ecosystems. This paper presents a framework for understanding and assessing these impacts. It is concluded that although much is not known, the impacts are likely to be wide ranging, resulting in both positive and negative effects.*

## **1. Introduction**

In its Second Assessment Report (SAR), the Intergovernmental Panel on Climate Change very briefly raised the issue of secondary or ancillary effects of climate change policies (Pearce, Cline *et al.* 1996, p.215). In 1997, the Intergovernmental Panel on Climatic Change released two technical papers addressing the implications of stabilizing atmospheric greenhouse gases and of proposed CO<sub>2</sub> emissions limitations (Houghton, Filho *et al.* 1997; Wigley, Jain *et al.* 1997). Unfortunately, no mention of ancillary impacts was made in either of these papers. In preparing its Third Assessment Report (TAR), however, the issue of ancillary impacts and the linkages between climate change policy and sustainable development, more generally, have received greater attention (Markandya and Halsnæs 1999; Munasinghe 2000).

In this paper, I focus specifically on the ancillary impacts, both positive and negative, on ecosystems from proposed GHG mitigation policies. Other papers and/or sessions in this workshop volume address issues more directly related to public health, transportation, agriculture, and land use. There will certainly be some overlap in these categories.

The concern about impacts of human actions on ecosystems preceded discussions of greenhouse gases and potential global climate change and has intensified in recent years for reasons that go well beyond this issue. Furthermore, most of the policies being proposed to address greenhouse gas emissions and atmospheric concentrations strongly resemble, if not duplicate, policies that have been proposed and/or implemented to address other issues, several of which have had ecosystem impacts as one of their principle concerns. Thus, it is reasonable to expect that climate change policies will have impacts on ecosystems.

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<sup>49</sup> Note, for example the recent EPA (1999) prospective study on the Clean Air Act Amendments of 1990. Based upon self-imposed criteria, including the ability to monetize flows of ecosystem services, the authors were able to identify only 5 endpoints for quantitative analysis, of which the monetary values for only 2 were included in their numeric results.

The framework I will use in addressing this issue draws from recent work in integrated environmental assessment and is shown in Figure 1 (blue or lightly shaded boxes). The GHG policies considered are examined for the *changes* they exert on the pressures ecosystems face. These lead to changes in the state of ecosystems, which result in both ecosystem and socio-economic impacts. Variations on this framework can be found in the literature, alternatively referred to as the Damage Function approach (Rowe, Smolinsky *et al.* 1995), the Impact Pathway approach (Holland, Berry *et al.* 1999), or the Pressure-State-Impact-Response approach (Rotmans, de Vries *et al.* 1997).

Figure 1 also shows the outline for this paper. Before addressing proposed GHG mitigation strategies and their potential impacts on ecosystems, it is necessary to provide some background on ecosystems – what they are, why they are of interest, and what are the principle causes of change in these systems. Finally, after addressing the potential impacts of GHG policies on ecosystems, I summarize the major findings of this study and its implications for climate change policy-making.

This procedure differs slightly from other studies, which have first defined major ecosystem types and then applied a similar approach to examine each of these (Watson, Zinyowera *et al.* 1996; Costanza, d'Arge *et al.* 1997; Watson, Zinyowera *et al.* 1998; US Environmental Protection Agency 1999). Given the nature and purpose of this paper, I have chosen not to do this, but rather to focus on the framework itself, citing particular examples without attempting to provide an exhaustive list of effects. It is hoped that the careful development and discussion of the framework will help to illuminate examples deserving of further exploration.

## **2. Background on Ecosystem Functioning, Goods, & Services – What are they and why are they of interest?**

There has been much discussion recently about ecosystem functioning, goods and services, and their value (de Groot 1992; 1994; 1997; Costanza, d'Arge *et al.* 1997; Daily 1997; 2000 #664; Pimental, Wilson *et al.* 1997; Heal 1999; 2000; Norberg 1999; Turner 1999; US Environmental Protection Agency 1999; de Groot, Van der Perk *et al.* 2000).<sup>50</sup> The terminology can be confusing. In this section, I discuss briefly one way to think about ecosystem functioning vis-à-vis goods and services and the values we can derive from these. For further detail on any of these topics, the reader is recommended to look at the referenced literature.

In their general functioning, ecosystems generate goods and services, from which humans derive value. Alternatively, we can state that, in the process of satisfying specific desires (values), humans draw upon goods and services that have been generated as a result of ecosystem functioning. Please note that, in general, there is not necessarily a one-to-one-to-one relationship between specific functions, goods and services, and values; specific values can be derived from a number of goods and services, which may rely on a number of functions.

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<sup>50</sup> Chapter 5 of the IPCC TAR Working Group II report, on Ecosystems and Their Services, is also addressing these issues.

Table 1 presents one classification of ecosystem functioning, goods, and services, developed by Gretchen Daily and colleagues as part of the “Ecosystem Services Framework” (Daily 1999). Table 2 translates these into values derived by humans from these goods and services.<sup>51</sup> I defer the important discussion of monetary/economic valuation until later in this paper, focusing here on simply cataloguing types of value. The categories in these tables represent just one way of slicing reality; other categorizations have been suggested (see references from first paragraph in this section). The specifics are less important than the following general points:

- Many, if not all, human needs and wants are met by ecosystem goods and services, directly or indirectly;
- Many of these goods and services cannot be substituted for, or, if so, not at reasonable costs;
- Functioning ecosystems are necessary for the continued provision of goods and services; and
- Human activities that impinge upon the ability of ecosystems to function will necessarily affect human welfare.

### 3. Background on Causes of Ecosystem Change

That human activities impact upon ecosystems in a myriad of ways should be of no surprise. Like all other species, we exist within the biophysical environment. What is relatively new, and of some concern, is that there exist very few remaining ecosystems that can be considered not dominated by humans (Vitousek, Mooney *et al.* 1997). Unfortunately, this domination is not always recognized, especially since much of our impact is unintentional. Furthermore, even where our impact is intentional, we rarely understand the full implications of our actions.

Table 3 presents a standard categorization of human pressures on landscapes and ecosystems, as specified by Rapport, *et al.* (1998). These are: harvesting, waste residuals, physical restructuring, magnified extreme events, and exotic species introductions. Harvesting represents the systematic removal of particular species from the ecosystem. It is of most concern when it occurs at high rates or affects keystone species. Waste residuals include both direct and indirect changes in the chemical environment. These can be in the form of air, water, or soil pollutants. Physical restructuring, or land transformation, can occur at small or large scales, and includes fragmentation of landscapes. It is considered the leading cause of the loss of biodiversity and other ecosystem goods and services worldwide (Vitousek, Mooney *et al.* 1997). Magnified extreme events are less well defined, but include such events as the large fires in the last decade, partially due to a century of no-burn policies that allowed fuel stocks to increase in forest ecosystems. Finally, exotic species introductions, whether intended or not, have the ability to reshape entire ecosystems.

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<sup>51</sup> Note that I have specifically excluded intrinsic value, which I define as an inherent value of an organism or ecosystem independent of human existence. As expressed in the preface, I view this, by its very nature, to be inaccessible to valuation by humans. This is not to be confused with existence value, which is the value that humans receive from knowing of an organism's or ecosystem's existence and is very much within the scope of valuation by humans. See Toman (1999) for a similar argument.



At this point, I am ready to address the specific question at hand. Referring back to Figure 1, the next section begins with a consideration of the policies being proposed to address the issue of atmospheric concentrations of GHGs, with an eye toward specifying the *pressures* they will generate. This will be followed by a discussion of the changes in *state* expected to result from these pressures. In particular, these will be examined in the context of the causes of ecosystem change outlined above. The next two sections will focus on the potential *impacts* of these changes, first considering the biophysical changes and then their socio-economic implications. The rubric of ecosystem functioning, goods and services, and value will structure these discussions.

#### **4. GHG Policies Considered (Pressures)**

From the perspective of reducing atmospheric concentrations of GHGs, there are two general options – reduce what is being added or increase what is being taken out. Of course, there are many alternatives for achieving these goals. In November of 1996, the IPCC released its first Technical Paper, which dealt with Technologies, Policies and Measures for Mitigating Climate Change (Watson, C. *et al.* 1996). For the TAR, the contribution of WGIII is to explore the issue of mitigation of climate change. Two chapters will specifically address the technological and economic potential of various mitigation options. Chapter 3 will focus on reducing emissions; Chapter 4 will focus on enhancing, maintaining and managing biological carbon reservoirs and geo-engineering. Finally, the Special Report on Land Use, Land-Use Change, and Forestry also addresses some of these issues (Watson, Noble *et al.* 2000).

Table 4 summarizes the principle alternatives being considered. These are divided between those that focus on the use of fossil fuels for energy production, those that focus on land-use practices, and those that fall out of either of these categories. The reason for this division will become clearer in the next section, where I indicate how these activities relate to the principle causes of ecosystem change discussed in the previous section. The last two alternatives have potentially large impacts on ecosystems, but they generally receive much less attention in the discussions of GHG mitigation, especially in the short-term. Population control is a much more fundamental and contentious issue; geo-engineering remains speculative and controversial. Thus, although these options are worth acknowledging, they are not dealt with further in this paper.

#### **5. Potential Non-Climatic Effects on Ecosystems (Changes in State)**

Is it possible to map GHG policies to causes of ecosystem change and, furthermore, to changes in the provision of goods and services? Ideally, this would be done for very specific cases, where the chains of causality could be identified in detail. This would require information not only on the options chosen, but also the areas of impact. This form of analysis has been attempted in several recent studies that examined the impacts of air pollutants (see, for example, Rowe, Chestnut *et al.* 1995; US Environmental Protection Agency 1997; Holland, Berry *et al.* 1999; US Environmental Protection Agency 1999). Here, I am more interested in providing a general framework, which may stimulate more detailed analysis.

Table 5 maps, in a general fashion, the linkages between the GHG policies considered and the principle causes of ecosystem change. Some specific examples are suggested in Table 6. Of course, even these need to be explored in the context of specific landscapes. Still, a number of general points can be highlighted:

- There is a wide range of potential effects;
- The greatest effects will likely be in the areas of waste residuals and physical restructuring;
- Changes in waste residuals will primarily result from changes in energy production and use; and
- Changes in physical structuring will primarily result from changes related to enhancing, maintaining and managing biological carbon reservoirs.

## **6. Ecosystem Changes (Impacts - Biophysical)**

Table 7 specifies a number of general changes to ecosystems and their provision of goods and services that can be expected from the above effects of GHG mitigation policies. This is a speculative list that would benefit from additional input. It does, however, give an indication of some of the key areas of concern for assessing the ancillary impacts of proposed policies on ecosystems. Furthermore, it indicates the importance of considering second and higher-order impacts and the fact that the impacts can be both positive and negative.

As pointed out earlier, a number of studies have organized the potential impacts by major ecosystem type (Watson, Zinyowera *et al.* 1996; Costanza, d'Arge *et al.* 1997; Watson, Zinyowera *et al.* 1998; US Environmental Protection Agency 1999). Others have been careful to note that the impacts must be considered at a number of biological/ecological scales, ranging from the molecular and cellular to global cycles (Norberg 1999; US Environmental Protection Agency 1999). A careful consideration from each of these perspectives would certainly help to illuminate further impacts.

Before moving on to the issue of the socio-economic implications of these changes, it is important to recognize how little we know about the actual physical nature of these changes. A number of recent compilations have attempted to shed light on these and related processes (see for example Daily (1997); Watson (1996); Watson (1998), US Environmental Protection Agency (1999), Levin (1999), and the special issue of *Ecological Economics* 29(2), 1999). However, most of the studies represented in these present only very general descriptions or are limited to very specific locales. It will require further advances in the underlying disciplines and their integration to truly understand these changes.

## **7. Socio-Economic Effects of Ecosystem Changes (Impacts – Socioeconomic)**

The next step in our analytical framework is to translate the biophysical impacts noted above into socio-economic impacts. An attempt to do so is shown in Table 8. Once again, this is a speculative list that would benefit greatly from further input. In particular, this list emphasizes the use value of ecosystem resources over their non-use values. Furthermore, our understanding of all impacts suffers from the uncertainties surrounding the functioning of ecosystems. However, the table does show that the potential impacts are wide ranging. It is also clear that there are both positive and negative effects that need to be considered.

When discussing socio-economic impacts, it is important to go beyond simply delineating the more or less tangible changes and link these to human values (see Table 2). Several of these – importance to human health, amenity value, productive use value, and consumptive use value – clearly fall out of the list of impacts in Table 8. The other values, albeit more subtle, cannot be ignored. The preservation and restoration of forest ecosystems certainly provides option value. The preservation of species most assuredly carries with it existence value.

The translation of these values into something meaningful for policy-making can perhaps be considered the ‘final’ step in our analytical framework. To do so, requires an introduction to the details of valuation in the context of ecosystems. This introduction will necessarily be brief. For further details on the general topic, the reader is referred to the many works in the literature on this topic (e.g. O’Connor and Spash (1999) Pearce (1993), Freeman (1993), Toman (1999), Costanza (1997), and Costanza (2000)).

Table 9, taken from Munasinghe (2000), lists a number of techniques that have been developed by economists to value ecosystem goods and services and the impacts of changes in their availability. DeGroot (1997) has, furthermore, tried to link the techniques to the values being considered to show which tools may theoretically be used to estimate different value. This is shown in Table 10.

Those methods that do not reflect actual behavior in conventional markets, i.e. those not listed in the upper left corner of Table 9, have been developed because, for the most part, ecosystem goods and services, among others, are not directly traded in conventional markets. The techniques have been applied in numerous studies and can be utilized to value the effects described in this paper. However, there remains large debate about the validity of these methods, particularly the contingent valuation method (CVM) (Arrow, Solow *et al.* 1993; Bateman and Willis 1999). As shown in Table 10, though, this presents a problem, as CVM is the only method that can be used to estimate a number of the values of interest.

In general, the limitations of these techniques have severely restricted the range of impacts that are included in analyses of human impacts on ecosystems (Rothman 2000). For example, based upon self-imposed criteria, including the ability to monetize flows of ecosystem services, the authors of the recent EPA (1999) prospective study on the Clean Air Act Amendments of 1990 were able to identify only 5 endpoints for quantitative analysis, of which the monetary values for only 2 were included in their numeric results.

Beyond the issue of not being able to capture many of the values of interest, a few other points need to be made concerning the valuation described above:

- These techniques focus almost exclusively on economic values, where the goal is economic efficiency. Other values, based upon concerns of fairness or sustainability are not well represented (Costanza and Folke 1997; Costanza 2000);
- The conceptual foundation for these forms of valuation is the value of scarce resources to *individuals*. For a number of ecosystem services, however, the values derived are better seen as community values (Toman 1999);
- These techniques are most appropriate for estimating the value of marginal changes in ecosystem services (Toman 1998; Heal 2000). As complex, adaptive systems, ecosystems may exhibit non-marginal effects from even marginal changes in pressures (see, for example Levin (1999));

- Many of these techniques are most appropriately applied to very specific locations/situations. There remain large problems with transferring these estimates to other situations and aggregating these (see, for example, Rothman (2000) and the special issue of *Water Resources Research* 23(3), 1992); and
- These techniques can be very time and cost intensive.

The criticisms leveled against these forms of economic valuation and the responses by their defendants has stimulated lengthy debates, but, in the eyes of many, little progress. There is recognition of the problems and inadequacies with these techniques, both in theory and practice. It remains a question, however, whether there exist any better tools.

There have been attempts to get beyond this impasse, however (see, for example, O'Connor (1998)). Here I wish to present examples of just two quantitative approaches that extend or complement economic valuation. Neither, to my knowledge, has been widely used.

The first was proposed by Bawa and Gadgil (1997) and is illustrated in Table 11. Their focus was on an assessment of the contribution of ecosystem services to subsistence economies, but the schema could be adapted for more general use. The second, proposed by Schneider, *et al.* (forthcoming), is referred to as the “five numeraires” (see Table 12).

Both of these build upon the strengths of economic valuation, but complement it with other approaches, where it is less appropriate.<sup>52</sup> The five numeraires approach, in particular, explicitly separates out the issue of loss of life from monetary valuation. If the amount of effort expended on applying and defending a dollar figure for the value of a (statistical) life were applied to other areas, it is likely that we would have a much better handle on the impacts of GHG policies and other areas of concern.

## 8. Summary

In this paper, I have laid out a framework to address the question of ancillary impacts on ecosystems from proposed GHG measures. The current state of knowledge does not permit a simple answer as to whether these are, on balance, positive or negative. It is apparent, though, that the impacts may be far ranging and significant.

It is my hope that further development and discussion of this framework will help to illuminate more examples of ancillary impacts deserving of further exploration. This, and advances in our understanding of ecosystems and their interactions with human systems are necessary to adequately answer the questions posed.

At the same time, I recognize that policy-makers cannot wait for definitive evidence to make decisions. By following the framework as defined, however, I feel that a better and more complete accounting of what we do know can be provided.

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<sup>52</sup> Inappropriate use of economic valuation has been at the core of the criticisms of two major efforts in the last decade –the impacts of climate change (Pearce, Cline *et al.* 1996) and the value of the world’s ecosystem services and natural capital (Costanza, d’Arge *et al.* 1997).

Table 1. **Ecosystem functioning, goods, and services**

<b>Production of Goods</b>
<i>Food:</i> terrestrial animal and plant products, forage, seafood, spice
<i>Pharmaceuticals:</i> medicinal products, precursors to synthetic pharmaceuticals
<i>Durable materials:</i> natural fiber, timber
<i>Energy:</i> biomass fuels, low-sediment water for hydropower
<i>Industrial products:</i> waxes, oils, fragrances, dyes, latex, rubber, precursors to many synthetic products
<i>Genetic resources:</i> intermediate goods that enhance the production of other goods
<b>Life-Support Processes</b>
<b>Regeneration Processes</b>
<i>Cycling and filtration processes:</i> detoxification and decomposition of wastes, generation and renewal of soil fertility, purification of air, purification of water
<i>Translocation processes:</i> dispersal of seeds necessary for revegetation, pollination of crops and natural vegetation
<b>Stabilizing Processes</b>
coastal and river channel stability
compensation of one species for another under varying conditions
control of the majority of potential pest species
carbon sequestration / partial stabilization of climate
regulation of hydrological cycle (mitigation of floods and droughts)
moderation of weather extremes (such as of temperature and wind)
<b>Life-Fulfilling Conditions</b>
aesthetic beauty
cultural, intellectual, and spiritual inspiration
existence value
scientific discovery
serenity
<b>Preservation of Options</b>
maintenance of the ecological components and systems needed for future supply of these goods and services and others awaiting discovery

Source: Daily 1999.

Table 2. **Values derived from ecosystem goods & services**

<b>Socio-Cultural Criteria</b>	<b>Short Description</b>
Importance To Human Health	These are derived from the maintenance of clean air, water, and soil, the development of new medicines, and the maintenance of mental health through the provision of opportunities for recreation and cognitive development.
Amenity Value	The direct enjoyment people derive from recreational activities in natural surroundings.
Heritage Value	The importance people attach to their cultural heritage, e.g., historic trees or specific landscape elements.
Bequest Value	The responsibility people feel towards future generations to conserve and enhance the evolution of natural ecosystems and biological diversity.
Existence Value *	The well-being ascribed to natural ecosystems and the wildlife they contain, as reflected in ethical and religious attitudes toward nature.
Option Value	The importance people place on a safe future (i.e., the future availability of a given amenity, good, or service) either within their own lifetime or for future generations.
Consumptive Use Value	The direct dependence of a community on natural ecosystems for resources (e.g., food), but also direct enjoyment of amenities (e.g., natural scenery)
Productive Use Value	The contribution of natural goods and services to economic production

\*I changed this from intrinsic value in the original, in keeping with my use of the terms existence and intrinsic value.

Source: modified from de Groot, 2000.

Table 3. **Human pressures on ecosystems and landscapes**

Harvesting
Waste Residuals
Physical Restructuring
Magnified Extreme Events
Exotic Species Introductions

Source: Modified from Rapport, Costanza et al. 1998.

Table 4. **GHG policies considered**

<b>Focused on Use of Fossil Fuels for Energy Production</b>
Curtailment of Energy Use Changes in Energy Extraction and Production Methods Improvements in Energy Efficiency Fuel Switching (including increased use of hydropower) GHG Capture
<b>Focused on Land-Use</b>
Increase or Maintain the Area of Land in Forests Manage Forests to Store More Carbon Manage Non-Forested Lands to Store More Carbon Reduce Dependence on Fossil Fuels through Product Substitution
<b>Other</b>
Geo-Engineering Population Control

Table 5. **Linking policies and pressures**

	<i>Pressure on Ecosystems</i>				
<b>GHG Policy</b>	Harvesting	Waste residuals	Physical restructuring	Magnified Extreme Events	Exotic Species Introductions
Curtailment of Energy Use		+			
Changes in Energy Extraction and Production Methods		+	++		
Improvements in Energy Efficiency		++			
Fuel Switching (including increased use of hydropower)	+	++	++		
GHG Capture		+	+		
Increase or Maintain the Area of Land in Forests		+	++		+
Manage Forests to Store More Carbon	+	+	++	+	+
Manage Non-Forested Lands to Store More Carbon	+	+	++	+	+
Reduce Dependence on Fossil Fuels through Product Substitution	+		+		
+: potentially small effects; ++: potentially large effects					

Source: author's interpretation.



Table 6. **Some specific non-climate effects on ecosystems**

Flooding of Landscapes for Hydropower Production
Conversion of Landscapes for Carbon Sequestration
Reduced Soil Erosion from Land Management Changes
Reduced Air Pollutants (Primary and Secondary) from Fossil Fuel Combustion
Reduced Air, Water and Toxics Pollution from Large-Scale Energy/Materials Extraction, Production, and Transport
Changes in Catastrophic Fire/Pest/Disease Potential in Heavily Managed Ecosystems

Table 7. **Some specific impacts on ecosystems (impacts - ecosystems)**

<b>Impacts on Biodiversity from Physical Restructuring</b>
Losses in Areas Flooded for Hydropower
Increases in Preserved and Restored Forested Areas
Losses in Areas Heavily Managed for Carbon Sequestration
<b>Other Impacts from Physical Restructuring</b>
Improvements in Net Primary Productivity from Reduced Erosion
Improvements in Water Quality from Reduced Erosion
Reduction in Flood Damage from Increased Water Retention and Reduced Erosion
Potential for Increased Pest/Disease/Fire Outbreaks
<b>Impacts on Local and Regional Ecosystems from Air Pollutants</b>
Reduced Eutrophication of Estuaries Associated with Airborne Nitrogen Deposition
Reduced Acidification of Freshwater Bodies Associated with Airborne Nitrogen and Sulfur Deposition
Improvements Tree Growth Associated with Damage from Ozone Exposure
Reduced Accumulation of Toxics in Freshwater Fisheries Associated with Airborne Toxics Exposure
Reduced Aesthetic Degradation of Forests Associated with Ozone and Airborne Toxics Deposition
<b>Impacts on Local Ecosystems from Large-Scale Energy/Materials Extraction, Production, and Transport</b>
Reductions in Damage from Oil Spills
Reductions in Damage from Strip and Underground Mining for Coal
Increased Bird Losses from Increased Wind Generation

Source: U.S. EPA 1999 and author's interpretation.

Table 8. **Some specific socio-economic effects (impacts - socio-economic)**

<b>Changes in Opportunities for Recreation</b>	
Changed Opportunities in Flooded Areas	
Changed Opportunities in Forest and Other Ecosystems	
Improved Opportunities on Freshwater Bodies	
<b>Provision of Consumables</b>	
Changes in Availability of Timber and Non-Timber Products from Forests	
Increases in Availability of Products from Freshwater Fisheries	
<b>Provision of Productive Inputs</b>	
Increases in Productivity of Forest Systems	
Increases in Productivity of Freshwater and Marine Systems	
Reductions in Expenses for Water Quality Treatment	
Changes in Availability of Biological/Genetic Resources	
<b>Other</b>	
Reduction in Damage to Health and Property from Flooding	
Potential for Increase in Damage to Health and Property from Pest/Disease/Fire Outbreaks	

Sources: U.S. EPA (1999) and author's interpretation.

Table 9. **Techniques for economic valuation of ecosystem goods & services and environmental impacts**

	<i>Type of Market</i>		
<b>Type of Behavior</b>	Conventional	Implicit	Constructed
Actual	Direct Purchases Effect on Production Effect on Health Defensive or Preventative Costs	Travel Cost Hedonic – wage, property value Proxy Marketed Goods	Artificial Market
Intended	Replacement Cost Shadow Project		Contingent Valuation

Source: Modified from Munasinghe (2000).

Table 10. **Techniques vis à vis values**

	<i>Value</i>							
<b>Technique</b>	Human Health	Amenity	Heritage	Bequest	Existence	Option	Consumptive Use	Productive use
Direct Purchase	x						x	x
Effect on Production								x
Effect on Health	x							
Defensive or Preventative Costs	x							x
Replacement Costs								x
Shadow Project						x		
Travel Cost		x					x	x
Hedonic – Wage, Property Value		x					x	x
Proxy Marketed Goods		x						
Artificial Market		x						
Contingent Valuation	x	x	x	x	x	x	x	x

Source: Modified from de Groot (1997).

Table 11. **Quantification of ecosystem services as proposed by Bawa and Gadghil**

	<i>Pressure on Ecosystems</i>				
<b>Parameter</b>	Direct Measure of Importance to Ecosystem People	Easy to Estimate	Estimates Non-Use Values	Incorporates Marginal Costs of Extraction and Benefits of Biodiversity	Importance to Policy Makers
Number of Persons Dependent on Ecosystem Services for Livelihood	x	x			
Value of Specific Products	x	x			x
Contribution to Cash Income	x	x			
Proportion of Households Dependent on Ecosystem Services for Livelihood		x			
Contribution to GDP			x		x
Value per Hectare			x		x

Source: Bawa and Gadghil (1997).

Table 12. **The five ‘numeraires’ as proposed by Schneider, et al.**

Monetary Loss Loss of Life Quality of Life (Including Coercion to Migrate, Conflict Over Resources, Cultural Diversity, Loss of Cultural Heritage Sites, etc.) Species or Biodiversity Loss Distribution/Equity
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Source: Schneider, et al. (forthcoming).

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# ESTIMATING ANCILLARY IMPACTS, BENEFITS AND COSTS OF GREENHOUSE GAS MITIGATION POLICIES IN THE TRANSPORT SECTOR

by Stef PROOST<sup>53</sup>

## 1. Introduction

This is a relevant topic because of three reasons. First there is a strong growth of CO<sub>2</sub> emissions in the transport sector. In the EU, it is expected that in 2010, the CO<sub>2</sub> emissions in the transport sector will be 40% higher than in 1990. In the US, growth rates could be 28% or higher [De Cicco & Mark, (1998)]. In developing countries and newly industrialised countries growth rates will be much higher as transport activity tends to grow faster than GDP in the industrialisation phase. These high growth rates make that the reduction of greenhouse gasses in the transport sector has become a priority for many policy analysts.

Second, there is no unanimity at all on the most appropriate policies to reduce GHG emissions in the transport sector. Mostly car use and air traffic are targeted but the type of policy instrument to be used remains unclear. Proposals include higher fuel taxes, speed limits, gas guzzler taxes on vehicles but also subsidies for mass transit.

Third, there are other important externalities in the transport sector (traffic accidents, congestion) and therefore the consideration of ancillary benefits could have a large impact on the policy choice.

It is not our intention to survey the whole field of transport and the environment<sup>54</sup>. Our aim is restricted to the analysis of policies that have been proposed to reduce GHG emissions. In section 2 we show on the conceptual level what are the ancillary benefits and costs that can be expected from different types of policies in the transport sector. It will become clear that the measurement of external costs of transportation is one of the key elements to determine ancillary benefits. The problems in estimating external congestion, air pollution and accident costs are dealt with in section 3. In section 4 we survey some studies that try to determine the costs of GHG reduction and the role of ancillary benefits. We distinguish between studies in the EU, the US and Developing Countries. In section 5 we conclude and sum up research priorities.

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<sup>53</sup> I thank the organizers A.Krupnick, D.Davis, my discussant, P.Crabbe and other participants of the workshop for their comments on a previous version of this paper.

<sup>54</sup> For a survey, readers can consult Bohm P., Hesselborn P-O, (2000) or Calthrop E., Proost S.(1998),

## 2. Conceptual issues: definition of private, external, social and ancillary benefits and costs of policies in the transport sector

We use a simple graphical model of the transport market to define the most important concepts on costs and benefits<sup>55</sup>. Later this illustration will be used to define the ancillary benefits of GHG reduction policies in the transport sector.

### 2.1 *A graphical approach to ancillary benefits*

We use a graph of one transportation market with two externalities: congestion and others (greenhouse gasses, air pollution, accidents etc.). We take congestion because it is the most difficult to understand and the most controversial. The transportation market we select is the use of a motorway between two cities during the peak period. We assume that the road infrastructure and the location of households and firms are fixed.

#### 2.1.1 *A transport market*

Consider the market for car km on a specific road link between two cities as depicted in Figure 1. This figure represents the market for car km in one particular period (peak) with one particular type of car (small petrol car with catalytic converter) on a road infrastructure with given capacity.

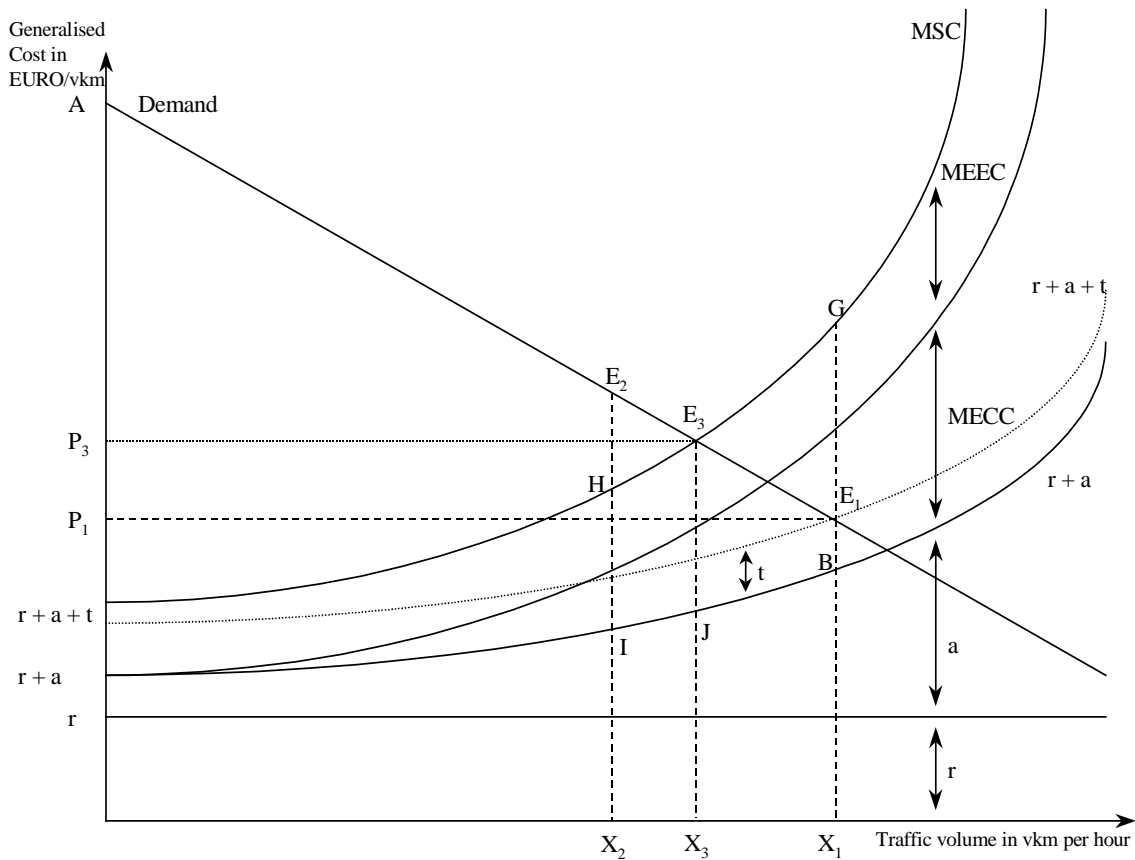
On the horizontal axis we represent the volume of car use (vehicle kilometre per hour). On the vertical axis we represent the generalised cost of car use. This generalised cost will equal the sum of the money cost (EURO/vehicle kilometre) paid by the car user plus the time cost needed per car kilometre.

The *demand function* expresses the marginal willingness to pay for car use at each volume of car-kilometres. The surface under the demand curve is thus a measure of the total benefits of car use: at a very high price only the strictly necessary car km would be demanded - as generalised costs drop, more and more households are ready to use the car for all types of purposes.

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<sup>55</sup> We use material from De Borger & Proost (2000) here. A more advanced treatment can be found in Small (1992)

Figure 1. A transport market



In this market, the volume of car use will be determined by the generalised cost of car use. Take any point on the vertical axis, the corresponding volume of car use on the horizontal axis is given by the demand curve: at this level of car use, the marginal willingness to pay of the last car user equals the generalised cost. Obviously, the volume of car use depends on many other elements as there are: prices of other modes and other goods, speeds and quality of other modes, location, income, composition and social attitudes of the household. All these other elements are kept fixed here.

In order to determine the equilibrium volume of car use we need to determine the cost for the user, we call this the *generalised private cost of car use*. The generalised private cost of car use consists of three elements: the resource costs, the taxes or subsidies and the average time cost. The resource costs equal the marginal production costs of the different inputs needed to use a car: fuel cost, maintenance cost, tyres and physical vehicle depreciation. It is represented by the line  $r$  in Figure 1. The resource costs plus average time costs are represented by the curve  $r+a$ . The average time cost increases when the volume of car use increases due to congestion: speeds drop and all drivers have higher time costs. When we add taxes on car use  $t$  (aggregate of taxes on fuel, maintenance, registration, etc.) we obtain the generalised private cost of car use (dotted line  $r+a+t$ ). In figure 1, this means that the equilibrium volume of car use is  $X_1$  and the generalised price equals  $P_1$ . This is the equilibrium we observe on the transport market.

### 2.1.2 External costs and taxation of transport

There can be external costs in this equilibrium. *External costs* are costs that are generated by a car user but not paid by him. The first externality is the marginal external congestion cost. The marginal external congestion cost is the cost of the additional time losses imposed on others by one extra car user.

This cost (MECC in Figure 1) is steeply increasing when we reach the capacity of the road network because of two reasons. First, adding one car decreases more and more the speed. Secondly, when there are more cars on the road, the decrease in speed will affect more cars. The marginal external congestion cost in Figure 1 corresponds to the increase in the average time cost curve times the volume of car use. It is important to recognise that, although every car user experiences congestion (higher time costs) himself, he does not pay for the time losses caused to other car users (the external part of the congestion costs).

We add a second external cost on top of external congestion costs: this can be air pollution, noise, accidents etc. (distance MEEC in Figure 1, taken more or less constant but this need not be the case).

The *total marginal social cost* of car use is now given by the sum of resource costs, average time costs, external congestion costs and other external costs. Taxes are excluded from the total marginal social cost. Taxes are a private cost but no cost at the level of society whenever taxes are returned to the households in an efficient way what we assume here. The marginal social cost includes all costs of car use. The optimal volume of car use would be reached when the marginal willingness to pay for the car use equals at least this social marginal cost. This means in Figure 1 that  $X_3$  is the optimal volume of car use: in this point the demand function (or marginal WTP curve) crosses the marginal social cost curve. The corresponding optimal generalised price equals  $P_3$ . This equilibrium can be reached by using an optimal tax  $E_3J$ . This tax equals the difference between the marginal social cost and the private cost of car use (before taxes). The efficiency gain of implementing this optimal tax equals the area  $E_3GE_1$ : the excess of social marginal costs over the value of car transport to the user as given by the demand function.

In the equilibrium shown in Figure 1, the total marginal external costs are only *internalised* partially: the tax paid per vehicle km is smaller than the total marginal external cost. The total tax paid per vehicle kilometre is too small. This is not the only problem. In general the tax paid is also not well tailored to the type of externality. This is important because there are different ways to decrease the level of externalities. First one can adapt the volume of car use as this affects the size of the external congestion cost but there is also the choice of vehicle type (more or less polluting), the driving style etc that all affect the size of the external air pollution, noise and accident costs. When a regulation forces all car drivers to use a cleaner car this will increase the manufacturing cost of cars ( $r$  will increase in Figure 1) but the size of the marginal external air pollution cost will decrease (MEEC in Figure 1 becomes smaller). A good air pollution regulation will make sure that the sum of the marginal external air pollution cost and the additional manufacturing cost of cars is as low as possible. When a good air pollution regulation decreases the marginal external air pollution costs, the optimal toll on car use decreases and the optimal level of car use could increase. This illustrates that policies affecting the volume of traffic (tolls, fuel taxes, road infrastructure,...) need to be coordinated with the policies affecting the type of vehicles used.

In our graphical example, the transport market had too low charges, this is a typical result for congested areas where the main tax policy instrument (fuel tax in the absence of time specific tolls) is unable to correct for the high external congestion costs. There exist many other transport markets (low congestion traffic on rural roads in countries with high fuel taxes) where the tax level is too high. In the latter case, car transport use is discouraged too much as in the equilibrium, the marginal WTP for extra trips is still higher than the marginal social cost.

One can raise the question *why we do not have a more efficient tax system* so that charges and taxes equal systematically the marginal external costs?. There are several reasons one can think of. A first explanation is the cost of a sophisticated tax system: making cars pay the proper external cost requires pricing differentiated by space and time, by driving style, vehicle type etc.. This is a very costly operation and therefore most countries resort to less expensive tax systems on fuels and vehicles that will overcharge some markets and undercharge other transport uses. A second reason is probably the complexity of the political decision process that makes that a growing problem like congestion is tackled too slowly because the construction of new roads and the increase in user charges are both unpopular decisions.

Figure 2. A transport market with capacity extension

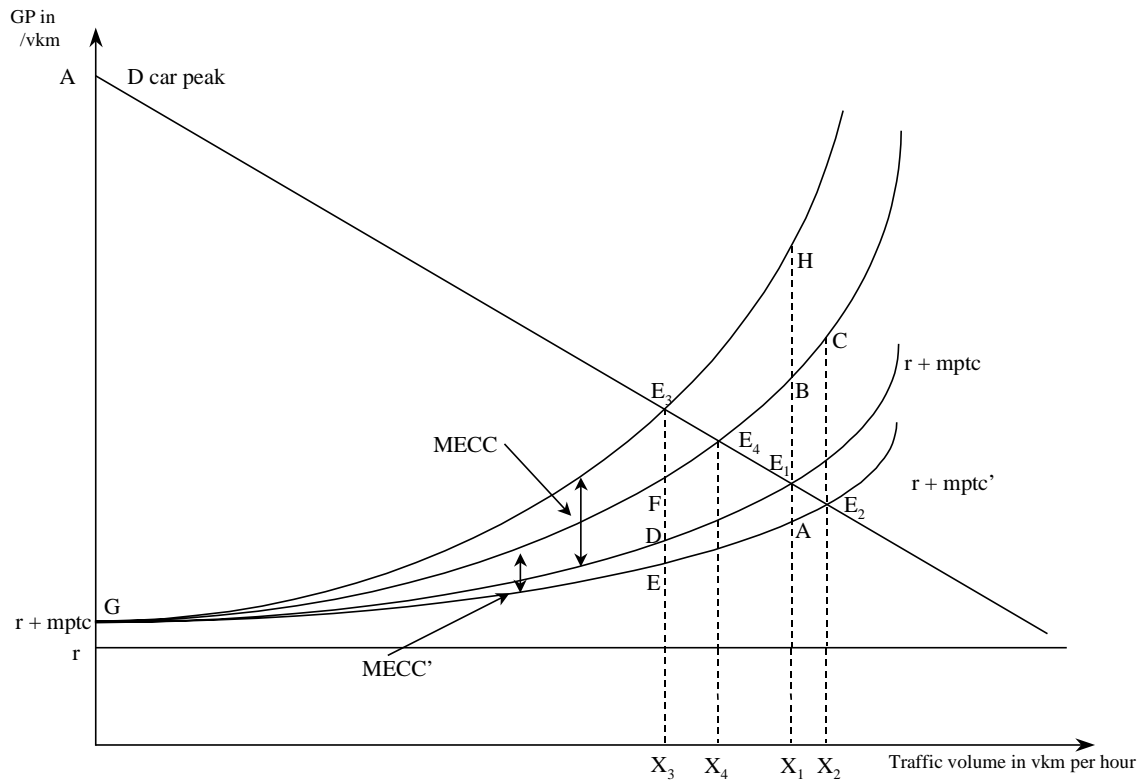


Figure 1 was constructed under the assumption that the road capacity was fixed. The road capacity determines the average time costs and is therefore an important policy variable to regulate the total quantity of transport. This is certainly the case in developing countries where the question is not to have any road extension or not but the pace at which the road network is extended and how to finance the investment. The effects of an extension of the road capacity can be discussed using Figure 2 that is of the same type as Figure 1. To simplify the exposition we assume that there are no taxes on car use. In the absence of taxes and before extension of the road capacity, the equilibrium was  $X_1$ . Important external congestion costs exist. We can now check what is the effect of a road extension. The extension of the road capacity means that the average time cost function and the private generalised cost (dotted curve in Figure 2) shift downwards as well as the marginal external congestion cost curve (dotted social marginal cost curve in Figure 2). The new equilibrium car use is now  $X_2$ . Note that speed is increased but the increase in speed is much less than expected as higher average speed attracts new traffic ( $X_2 > X_1$ ) because the generalised price went down. It would have been better if there had been no increase in traffic as the newly generated traffic decreases total economic efficiency by the area  $BCE_2A$ . The net benefits of road extension will be the decrease in social costs for the existing traffic (area  $GHB$ ) minus the net efficiency costs of induced traffic ( $BCE_2A$ ). This net benefit has to be computed for every future year and the discounted sum of these net benefits can be compared to the investment cost including external environmental costs associate to the infrastructure construction. When the net benefit is larger the investment is economically efficient.

It is important to realise that different investment decisions need to be taken when pricing of traffic is more efficient. Starting with optimal pricing in equilibrium  $X_3$  the same road extension would now lead to the new equilibrium  $X_4$ . Again new traffic is generated but the optimal pricing policy makes sure that this new traffic is justified from an efficiency point of view: the marginal WTP of the attracted traffic is larger than the social marginal cost. The net benefit of a road capacity extension is now smaller:  $GE_3 F + E_3 E_4 F$ . In developing countries where the growth of demand is high, infrastructure extension is an important component of transport policy. The need for infrastructure extension and the corresponding induced demand reactions can be contained if an effective pricing policy is pursued. Correct pricing of road use tends to reduce the need for infrastructure extension.

A frequent question is whether external cost pricing (or short run marginal cost pricing) will cover the investment costs? If investment policy is optimal, and if tolls equal at least the marginal external congestion costs, the revenue of the toll will equal the marginal infrastructure extension cost. If this cost is constant, tolls will at least pay for the investment cost. This also implies that, given optimal pricing and investment, the level of congestion is not zero<sup>56</sup>.

## 2.2 *Policies to reduce GHG emissions and the definition of Ancillary benefits*

Ultimately we are interested in computing the welfare costs of CO<sub>2</sub> reduction policies in the transport sector. The different costs of policies can be illustrated using Figure 1 and 2. We discuss briefly the following policies: vehicle fuel efficiency standard, fuel tax, transit subsidy, transport infrastructure policies and location policies. For each of these policies we describe the expected effects and the costs and benefits (excl. climate change benefits).

*We define ancillary benefits as the benefits of greenhouse gas reduction policies other than the climate change benefits.* This definition only makes sense when we know what is included in the costs of a greenhouse policy. We define as cost the direct resource costs of the emission reduction policy that the economic agents have to bear. Using this type of definition means that ancillary benefits will only exist whenever there exist non-internalised externalities other than climate change [Markandya, Krupnick, Burtraw, (2000)]<sup>57</sup>. We will use this definition throughout this text. There exist other definitions but discussing them would not be very interesting. In the end what matters is to include in the net cost of GHG reduction policies, all costs and benefits associated to this policy other than the Climate Change benefits themselves. This is the basic requirement for any Cost-Benefit Analysis.

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<sup>56</sup> For a pioneering numerical illustration of the relation between investment and pricing one can consult Keeler and Small (1977).

<sup>57</sup> We refer to Markandya, Krupnick, Burtraw, (2000) for a more thorough treatment of second order effects. One of these important second order effects are the tax recycling effects that can be particularly important in the (highly taxed) transport sector. Interested readers can consult Parry and Bento (1999) and Mayeres and Proost (1997 and 2000).

### 2.2.1 Vehicle fuel efficiency standard

This is one of the most frequently used policies. A fuel efficiency standard will make new cars relatively more expensive<sup>58</sup>, in the long term this will increase the costs of car use. In Figure 1, line r will increase but the private generalised cost (r+a+t) will increase less when there are important fuel taxes because the more fuel efficient car saves also fuel taxes. In total, this will result in lower car use and smaller CO<sub>2</sub> emissions per vehicle kilometre

The costs and ancillary benefits of this policy are summarised in Table 1.

**Table 1. Costs and ancillary benefits of vehicle fuel efficiency standard**

	<i>Suppressed traffic</i>	<i>Remaining traffic</i>
Costs of policy	<ul style="list-style-type: none"> <li>• Lost consumer surplus</li> <li>• Lost tax revenue</li> </ul>	<ul style="list-style-type: none"> <li>• Increase in resource costs of vehicle use</li> </ul>
Ancillary benefits	<ul style="list-style-type: none"> <li>• Saving in external air pollution, noise and accident costs</li> <li>• Saving in external congestion costs (= reduced time costs of remaining traffic)</li> </ul>	<ul style="list-style-type: none"> <li>• Decrease in external air pollution costs</li> </ul>

The most important ancillary benefits will probably be the savings in air pollution costs (other than CC) for the remaining traffic and the savings in external congestion costs through the reduction in the traffic level. This means that the effect on traffic volume will be one of the major determinants for ancillary benefits. Ancillary benefits correspond here to savings in external costs (other than climate change). If external costs would have been perfectly internalised by taxes, the lost tax revenue for suppressed traffic will equal the savings in external costs for suppressed traffic and Table 1 is simplified. The only ancillary benefit that remains is the decrease in external air pollution costs. In general taxes will not internalise external effects perfectly and will overshoot external effects on some markets and be lower than external effects on other markets.

### 2.3 Fuel tax policy

Conceptually this policy is close to a fuel efficiency policy but the order of magnitude of the different effects will be different. To save the same total quantity of CO<sub>2</sub> as with a fuel efficiency standard, a fuel tax policy will count less on the improvement of the fuel efficiency and rely more on the reduction in the volume of traffic. The reason is that the car driver now also pays more for the remaining fuel use. This leads to costs and ancillary benefits that are different from those of a fuel efficiency standard. Compared to table 1, the suppressed traffic effect becomes more important so that the ancillary benefits will consist more in saved external congestion costs than in saved air pollution costs.

<sup>58</sup> We assume a proper functioning of the car market so that adding an extra technical requirement can only increase the price of a car.



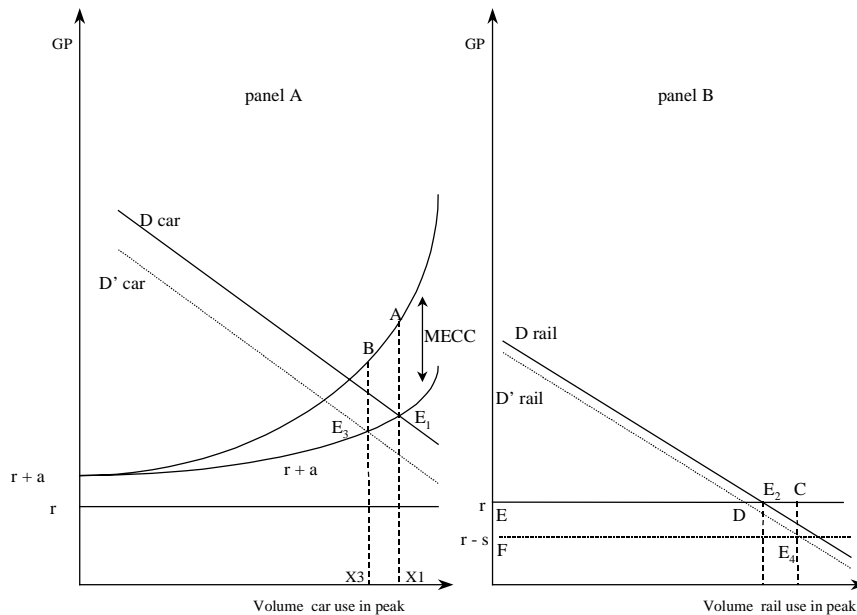
## 2.4 Public transport subsidy

The subsidy to public transport as a greenhouse gas reduction policy is in general motivated by the better fuel efficiency per passenger km of public transport.

The interactions to be taken into account are illustrated in Figure 3. We start in Panel A of this figure with a given volume of car use  $X_1$  that is too large: there is an important marginal external congestion cost (A  $E_1$ ). In Panel B we have a rail service where the price equals the marginal cost  $r$ . The equilibrium is  $E_2$ . We can simulate the effects of a subsidy  $s$  to rail in Figure 3. The subsidy decreases the price of the rail mode to  $r-s$ . This will make the demand curve for car use shift to the left ( $D'$ ): for the same generalised cost of car use there will be less car users because some of them prefer the train. When taxes on the car market remain unchanged (to keep it simple we have assumed no taxes here), the external congestion cost decreases to  $BE_3$ . Because the equilibrium volume of car use decreases to  $X_3$  there will be a decrease of the generalised cost of car use (the average time cost decreases). The decrease in the generalised cost of car use will produce a shift to the left of the demand function of rail use ( $D'$ ). The ultimate equilibrium is  $E_3$  for car use and  $E_4$  for rail use.

In order to compute the net welfare gain of this subsidy one needs to balance the welfare loss on the rail market with the welfare gain on the car market. There is an efficiency loss on the rail market because some users now make trips that do not cover the marginal resource cost of rail trips. There is a welfare gain on the peak car market because the number of car trips for which the willingness to pay is lower than the social marginal cost has been reduced.

Figure 3. Effects of subsidies to public transport



The costs and ancillary benefits are summarised in Table 2.

**Table 2. Costs and ancillary benefits of a public transport**

	<i>Rail market</i>	<i>Car market</i>
Cost of policy	<ul style="list-style-type: none"> <li>• Efficiency loss on rail market</li> </ul>	
Ancillary Benefits	<ul style="list-style-type: none"> <li>• Increased external costs of rail market</li> </ul>	<ul style="list-style-type: none"> <li>• Saving in external air pollution , noise and accident costs due to suppressed volume</li> <li>• Saving in external congestion costs (= reduced time costs of remaining traffic) due to suppressed volume</li> </ul>

### **2.5 *Modal shift policies in the freight sector***

The idea is similar to the subsidies to public transportation. Now the subsidies are given to modes like rail and inland waterways that have in general lower GHG emissions per ton km transported than trucks and airplanes. As many of these markets have important external effects, the ancillary benefits or costs can be important.

### **2.6 *Road investment policy***

Not to extend roads can be considered as an instrument to contain the growth of traffic and to reduce the emission of GHG (cf. our Figure 2). Compared to the situation with road capacity extension, there will be costs and ancillary benefits associated to the remaining traffic and the suppressed traffic. There will be suppressed traffic and remaining traffic and the costs and ancillary benefits are listed in Table 3.

**Table 3. Costs and ancillary benefits of not extending the road capacity**

	<i>Suppressed traffic</i>	<i>Remaining traffic</i>
Costs of policy	<ul style="list-style-type: none"> <li>• Lost consumer surplus</li> <li>• Lost tax revenue</li> <li>• Saved investment cost (benefit)</li> </ul>	<ul style="list-style-type: none"> <li>• Increase in average time costs</li> </ul>
Ancillary benefits	<ul style="list-style-type: none"> <li>• Saving in external air pollution , noise and accident costs</li> <li>• Saving in external congestion costs (associated to average time cost function with road extension)</li> </ul>	<ul style="list-style-type: none"> <li>• Increase in other external air pollution costs ?</li> </ul>

### 2.6.1 *Location policy*

Transportation is the result of passenger trips between home, school, job and leisure locations and of freight transport between the producer and user of inputs and outputs. The most obvious way to reduce CO<sub>2</sub> emissions, is to reduce the need for transport flows through relocation of activities. This idea looks simple but analysing its implications fully is a complex undertaking. The problem is that higher concentration of activities saves emissions but can also imply economic costs. These can consist of loss of specialisation (higher overall production costs) and of an increase in other external costs. Noise, air pollution and industrial risk impacts can be larger in more concentrated locations.. In a recent survey on urban spatial structure, Anas, Arnott and Small (1998) find that urban economics has not yet clear views about the determinants of city size and optimal city planning.

Location policy is a potentially very important instrument; certainly in developing countries where urban growth rates are high. A minimum requirement for a good location policy is that there is close coordination in land use policy and in the construction of public transport capacity. Light rail or metro systems only make sense for very high densities of population.

## **3. Problems in the estimation of ancillary benefits**

Ancillary benefits of GHG reduction policies in the transport sector will consist mainly of two types: time savings for remaining road traffic due to a decrease in road transport volume and savings of external costs (other than congestion) due to either a reduction in transport volume or due to a decrease in the intensity of external costs per vehicle kilometre for remaining transport flows.

We discuss briefly<sup>59</sup> the estimation problems for congestion gains, traffic accidents, and conventional air pollution. We add a fourth problem: the treatment of resource costs that are not paid by the user.

### **3.1 *Reduction of congestion***

Any reduction of transport volume for a congested mode brings extra time benefits for the existing users. Estimating these benefits raises two issues.

The first is the estimation of the *value of time*. This issue was important for transport experts and nowadays there exist many studies using revealed preference and stated preference techniques. They give a range of time values for different trip purposes and comfort conditions. Time values differ according to purpose, comfort, income level etc.. Although there exist wide differences in time values, these differences can be rationalised and values of time should not be considered as a major difficulty.

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<sup>59</sup> More elaborate discussions of marginal external costs can be found in Jansson (1994), Calthrop & Proost (1998), Green, Jones, Delucchi (1997), Mayeres, Ochelen, Proost (1996), Maddison *et al.* (1996).

The second problem is the inclusion or not of *schedule delay costs*. There are two competing formulations for a congestion cost model. The first model uses a speed-flow relationship where adding extra traffic volume decreases average speed that can be translated into time losses. The second model is the bottleneck model (Arnott, De Palma, Lindsey (1993)) where the peak period is of variable length: once the road capacity is reached, drivers incur queuing costs (time losses) but also schedule delay costs. The second type of model will tend to generate much higher congestion costs. Of interest is that most formulations tend to use rather the first type of formulation and could therefore underestimate the congestion problems.

### 3.2 *Traffic accidents*

The external accident costs are probably the most controversial topic in the estimation of the ancillary benefits. Imagine that we are able to reduce car use. Are there any savings in traffic accident costs to be included in the ancillary benefits? There are two sources of benefits (or savings in external costs): first the reduction of external accident costs for constant accident risks and secondly the change in accident costs due to the change in accident risks.

When we keep the average accident risk constant, a reduction in the volume of traffic will save accident costs. This can only be considered as a net benefit if the driver did not already take these into account. A driver takes into account the accident costs by two mechanisms: he takes into account his own accident costs (including the valuation of relatives and friends for his loss of life or injury) and his insurance premium. If his insurance premium covers all average accident costs and is related to his driving decisions, the average accident costs are taken into account by the user. Insurance premiums do probably not pay for all accident costs: some “cold blooded” costs as there are police costs, medical emergency services etc. are probably not paid and it is not clear what type of subjective value of life and injuries<sup>60</sup> is taken into account. Secondly, insurance premiums are mostly an annual payment unrelated to the number of km driven. The “pay at the pump” advocates conclude that insurance premia are not taken into account at all by drivers (Kavalec & Woods (1999)). This is not fully correct: annual insurance payments still determine the car ownership decisions and more and more insurance contracts link the premium to the personal accident record and therefore to the annual mileage. Obviously, if there is no car insurance at all as is probably the case in some developing countries, traffic accident costs can be an important component of ancillary benefits.

Assume from now on that the average accident cost is paid but that the average accident risk increases with the traffic volume. In this case drivers pay average costs and not marginal costs and any reduction of traffic volume generates an ancillary benefit equal to the difference between marginal and average accident cost. Initially, several authors (Vickrey,(1963), Newbery (1990)) used a model with this feature. Recent empirical studies of the relationship between traffic volumes and accident risks (Dickerson *et al.*(2000). show that average accident risks stay more or less constant at low to medium traffic flows and increase at high capacity utilisation rates, an externality may exist there but this still needs to be corrected for differential impacts of the volume of traffic on the type of accidents. Accidents may become less severe at high congestion levels.

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<sup>60</sup> The estimation of the subjective value of loss of life and limb remains an important research topic in economics as the traditional techniques (CVM, RP) don't work that well

### 3.3 *Air pollution*

Air pollution is a traditional example of external costs. The main difficulties in estimating saved air pollution costs are the dose response relationship and the estimation of the value of years of life lost and of health problems. This issue is treated in depth in a companion paper for this symposium: Krupnick, Davis and Thurston (2000).

### 3.4 *Other unpaid resource costs*

Most studies assume implicitly that all traditional resource costs (for a car: car, fuel, maintenance, parking and so on) are paid by the car user. This need not be true. For cars a common counterexample is free parking offered by employers or shopping centres. Whenever more employees take mass transit to go to work or more people go shopping by bus as part of a GHG reduction policy, there is a saving of parking resource costs that could be counted as ancillary benefit. The story is more complicated than this. Making people pay for their parking costs may actually increase distortions on the labour market (because one discourages labour supply even more) and there may be high transaction costs associated to billing for parking.

Mass transit raises other challenges. In some countries, users don't pay anything or less than the marginal social cost. The marginal social cost may itself also be difficult to compute because of economics of density in public transport [Small (1992)].

## 4. **Climate change policy studies in transport: how important are the ancillary benefits?**

We review some of the existing studies by geographical area. An approach by area is needed because there are major differences in the present transport policies. As we had an easier access to recent unpublished European studies they receive more emphasis. This could also reflect a stronger interest in GHG emissions in the transport sector in Europe compared to the US. Very few studies are available for the Developing Countries. It is probably in these countries that exist the highest needs for transport and environment policy studies.

### 4.1 *Europe*

European transport policies are characterised by high densities in urban areas, relatively low mobility, high fuel taxes and a well-developed system of mass transit (rail, metro, bus). The last 5 years, the European Commission has been advocating the use of better transport pricing policies. Different European research consortia (PETS, TRENEN-II, AFFORD) have studied the potential benefits of marginal social cost pricing. These projects together with a study by Koopman (1995) will be our main sources.

In the TRENEN II consortium (De Borger and Proost 2000), the expected private costs of car use and of other modes are compared to the social marginal cost of using these modes. This is the type of information we need to determine in what type of transport equilibrium we are now (in terms of Figure 1: are we in equilibrium  $X_1$  or  $X_3$ ?). The comparison of private prices and social costs will tell us also what are the major types of non-internalised external costs and these are at the origin of ancillary benefits. The social marginal cost includes all resource costs together with the external cost of congestion, accidents, noise, climate change and other air pollutants. The external cost of air pollution was extrapolated from EXTERN-E results (Bickel *et al.*, (1998)) and includes climate change benefits. Figures 4 and 5 compare for different cities and non urban areas, the cost per car kilometre of a private user that does not have to pay for his parking spot (most drivers don't) and the marginal social cost expected for 2005 when policies are unchanged. Figure 4 deals with the peak period. For each area, two bar charts are shown. The first bar represents the private car user costs that consist of the sum of the resource cost (production cost of car, maintenance and fuel cost), the price of parking (zero by assumption here), the taxes and the time cost. The second bar chart represents the marginal social cost of car use that consists of the sum of resource costs, parking resource costs and the marginal external costs. Figure 4 shows that there is an important discrepancy between the private users' price (left bars) and the social marginal cost (right bars) in the peak period for cars. The major problems are the unpaid resource cost of parking in urban areas and the external congestion costs. Similar graphs exist for public transport and freight transport. Almost all modes of transport are underpriced in the peak, some of them because of the very high external costs, others because they are heavily subsidised. The discrepancy is much less pronounced in the off peak period where car use is sometimes overtaxed.

Figure 4. Peak car reference prices and costs (expected for 2005)

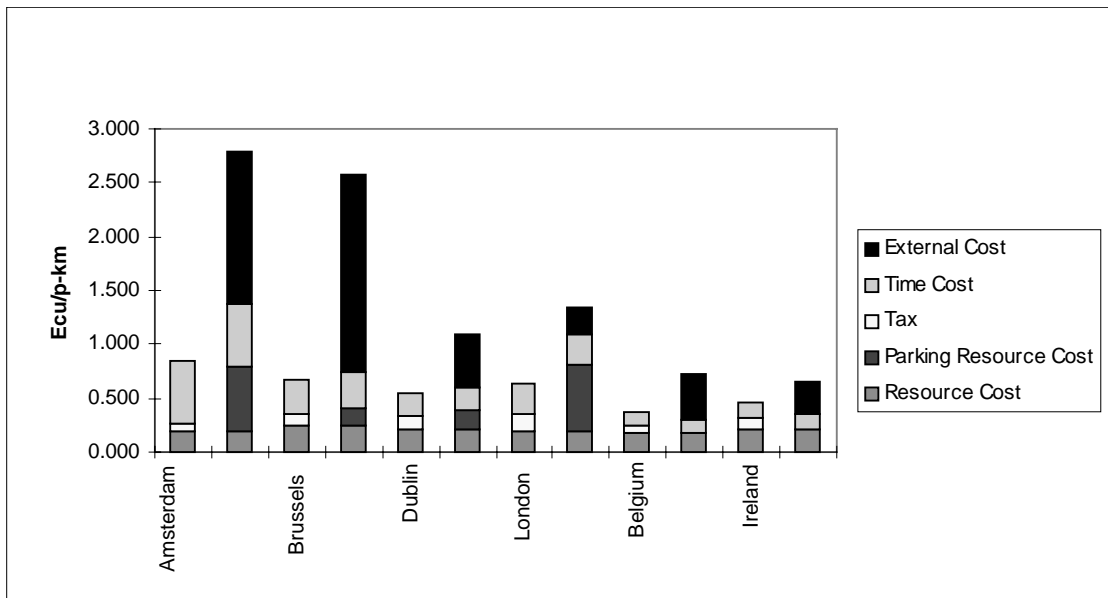
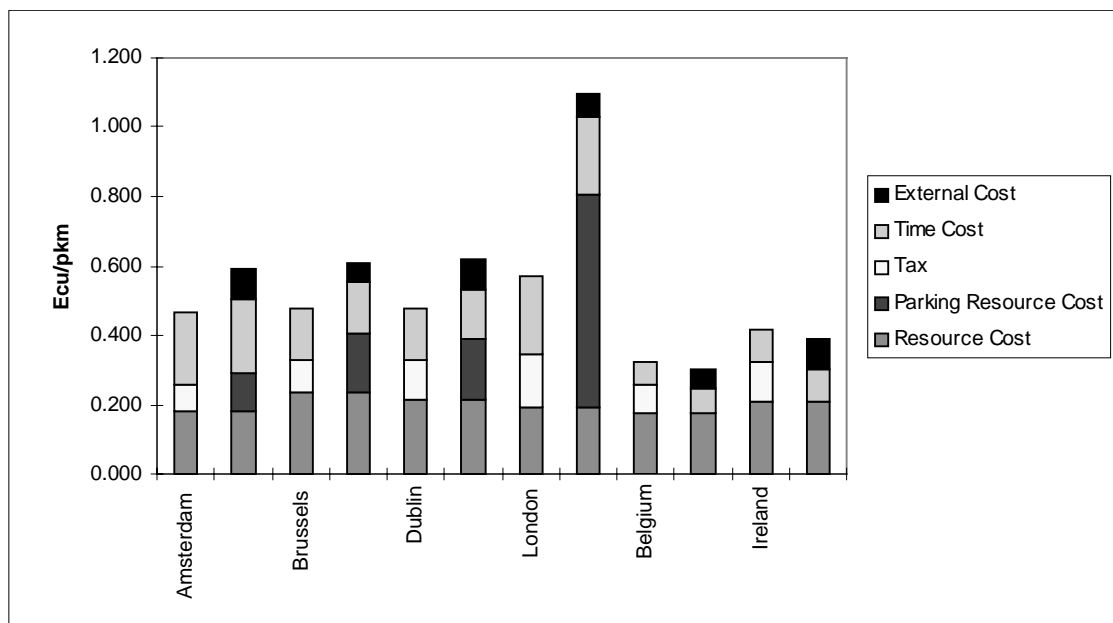


Figure 5. Off-peak car reference prices and costs (expected for 2005)



We analyse here more in particular the case of Brussels. The structure of the marginal external costs is given in Table 4 (Proost & Van Dender (1998)).

Table 4. Structure of marginal external costs for a small car in Brussels in 2005

in EURO/Vehkm	Gasoline		Diesel	
	Peak	Off peak	Peak	Off peak
Air pollution	0.004	0.004	0.042	0.026
Accidents	0.033	0.033	0.033	0.033
Noise	0.002	0.008	0.002	0.008
Congestion	1.856	0.003	1.856	0.003
<b>Total</b>	<b>1.895</b>	<b>0.047</b>	<b>1.932</b>	<b>0.068</b>
Tax	0.12	0.11	0.08	0.07

One can see that the price inefficiencies are dominated by external congestion costs that only appear in the peak period and that, as regards air pollution, diesel is the major problem because of the health problems attributed to PM<sub>10</sub>. The low air pollution costs are the result of the implementation (by 2005) of many recently decided regulations on car emissions in the EU. For CO<sub>2</sub> a damage estimate of 25 EURO/ton of CO<sub>2</sub> is used. Appropriate instruments can probably reduce each of the external costs but it is already clear that the congestion issue will drive most policy assessments.

In the end, the inefficient transport market is the result of wrong tax and pricing policies. The TRENEN – II model can be used to look for a welfare optimum for any given set of policy instruments. In Table 5, taken from Proost and Van Dender (1998), the effects of different policy options are compared. The first column of this table reports the net economic efficiency effect: this equals the sum of:

- changes in generalised consumer surplus (contains value of changes in time costs) and this for all markets (except labour);
- changes in producer surplus;
- changes in air pollution costs, noise costs and external accident costs;
- changes in tax revenue that received a small premium (7%) to account for the efficiency effects of using the tax revenue to reduce labour taxes.

The efficiency gain obtained with perfect pricing is used as benchmark for the other policy instruments. The three other columns report different effects: change in air pollution damage, total volume of car transport and average speed in the peak period.

**Table 5. Global efficiency of alternative transport and environment policy instruments for Brussels in 2005**

	<b>Change in economic efficiency (mio EURO/day)</b>	<b>Change in air pollution damage (mio EURO/day)</b>	<b>Total volume of passenger car units</b>	<b>Speed of cars in peak (km/h)</b>
Reference	0		100	23
Perfect marginal social cost pricing	100% (= +0.703)	-0.015	78	40
Cordon pricing	+52%	-0.001	89	33
Parking charges	+32%	-0.005	95	26
Emission standard (consumer paid)	-0 %	-0.006	100	23
Emission standard (government paid)	-0%	-0.006	100	23
fuel efficiency standard (consumer paid)	-17%	-0.016	98	24
fuel efficiency policy (via fuel tax)	+5%	-0.016	95	26



Perfect pricing of external costs leads to lower air pollution damage mainly as side effect of lower volume of car use. The lower value of car use is the result of different effects that are mainly targeted at reducing congestion: more car pooling, switch to other modes and a smaller number of trips. This table illustrates that the welfare maximising policies for the transport sector are those policies that address as directly as possible the problem of congestion and unpaid parking. The air pollution benefits of this policy (-0.015) are only 2% of the total efficiency gains that are achieved in this scenario. These benefits are the result of smaller volumes (passenger km decreases by 22%, carkilometre by more than 30%) and of a smaller share of diesel cars.

Congestion problems can be tackled by cordon pricing (toll levied on commuters at entrance of city, the toll is differentiated between peak and off peak) or by parking charges. In the parking charges policy, all drivers are forced to pay for their parking costs (at destination), moreover the parking charges contain a special tax to discourage the overall level of car use. Both policy instruments generate important efficiency gains. The size of the efficiency improvement is strongly correlated to the increase in speed they can generate in the peak period.

The emission standard scenario assumes that one can get cars with lower emissions of conventional pollutants at an investment cost per vehicle that varies between 225 and 824 EURO per car. These are data taken from the AOP-I results. The efficiency benefits vary slightly in function of whether the consumer or the government pays for the cleaner cars. There is a difference because government funds have a marginal cost higher than one (in fact 1.07) and because there is an income effect for the consumer that affects demand for transport. Such emission standards can give rise to important reductions in the emission of conventional pollutants but the total efficiency gain is smaller and even negative. The explanation lies in the high marginal abatement cost that is not compensated by air pollution benefits.

The fuel efficiency standard scenario corresponds to the introduction of the 5 litre car in 2005. The second fuel efficiency scenario means that the use of a 5 litre car is stimulated via higher fuel taxes rather than through a standard imposed by government. Both scenarios generate approximately the same gain in air pollution benefits. These air pollution benefits consist mainly in the reduction in diesel fuel and in the lower emissions of PM<sub>10</sub>. The fuel efficiency standard is a less interesting policy than the fuel tax policy because in the former there is almost no effect on the volume of transport and on congestion.

Not everybody shares the view that fuel efficiency standards are a very costly option to reduce CO<sub>2</sub> emissions. In the EU, the major policy decision on CO<sub>2</sub> emissions is the voluntary agreement on fuel efficiency standards that is concluded with the association of automobile manufacturers. The proponents of fuel efficiency standards point to the benefits for myopic consumers<sup>61</sup> that are not aware of the fuel costs and to the large technological potential. The major flaw in their argumentation is that the present high excises on gasoline and diesel fuel make that the marginal cost of making more fuel efficient cars is indeed low for consumers. They use consumer prices to estimate the benefit of one litre of fuel saved. From a society point of view prices before tax have to be used to compute the real benefits of more fuel efficient cars and this reduces these benefits to one third or less.

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<sup>61</sup> Koopman (1995) using the EURCARS model of the European Commission finds that a CAFE standard is only slightly more costly than a CO<sub>2</sub> tax for the same 10% reduction in CO<sub>2</sub> emissions in the transport sector. His result can be explained by the very high implicit discount rate (up to 50%) he attributes to car buyers. There is no clear empirical evidence for such a high rate. Other studies of the car market (Verboven, 1997) point to a more normal 10%.

Table 5 is an illustration for one urban area in Europe. Part of the analysis has been redone for other areas. The major conclusions are that off-peak road traffic tends to be taxed too much and that peak road transport in urban areas is underpriced [De Borger & Proost, (2000)].

Table 5 is useful to illustrate the relative importance of different types of *ancillary benefits* and their impact on the policy selection process. The most important external cost and potential ancillary benefit is probably congestion in urban areas. The traditional instruments for GHG reduction in the transport sector (fuel efficiency policies and fuel tax policies) are not very cost-effective and generate almost no ancillary benefits. The reasons are that the fuel price instrument has already been used too much and is not time and place specific and that the existing air pollution regulation is starting to put very clean (conventional pollutants) on the market. New instruments (road pricing, parking charges) that affect the congestion problems in a more direct way can generate important overall efficiency gains and reduce the emission of GHG as a by product. These policies could be considered as GHG policies with very high ancillary benefits.

## 4.2 USA

In the USA, fuel efficiency policies have been used in the past and it looks as if they are the major instrument considered to save GHG emissions in the transport sector. Bernow and Duckworth (1998) count on mainly fuel efficiency policies to stabilise GHG emissions in the transport sector between 1990 and 2010. For cars they count on a fuel efficiency improvement of 1mpg per year, reducing the average consumption from 25 mpg (9.4 l/100km) to 45 mpg (5.2 l/100km) in 2010. After 2010, one counts on new fuels and new vehicles to improve the fuel efficiency.

Originally, fuel efficiency policies have been introduced to reduce oil import dependency and not air pollution emissions. The CAFE (Corporate Average Fuel Efficiency) policy has been studied extensively. Green and Duleep (1993) and Greene (1998) show that the CAFE regulation succeeded in bringing down the fuel consumption by cars at a low cost. The major benefits are fuel savings (if oil prices continue to increase and if discount rate is low) and oil market effects (the international oil price decreases and the security of supply improves through a leftward shift of the demand curve). The major cost is the increase in manufacturing costs of cars<sup>62</sup>.

Ancillary benefits (or costs) of this type of GHG reduction policies (beyond the oil market effects) are the effects on the emissions of other air pollutants and effects on traffic safety. The effects on congestion will be small as the overall car use was almost not affected. CAFE standards on cars could have deteriorated the urban air quality by increasing the life of older vehicles and by a shift to unregulated light trucks. According to Green (1998), these effects exist but are not that important. CAFE standards reduce fuel consumption and indirectly also the emissions of other pollutants. Harrington (1997) has shown that, although for new cars there is no relationship between tailpipe emissions and fuel consumption, there is a close positive correlation between fuel consumption and VOC and HC emissions for older cars.

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<sup>62</sup> Knowing the cost of emission regulation of cars is far from obvious [see Bresnahan, Yao (1985) and McConnell *et al.* (1995)].

There are two ways a CAFE standard can affect traffic safety. It can affect the overall volume of car use and it can affect the type of car that is build. The overall car use was almost not affected. When more fuel efficient cars means lower vehicle weight, the fatality rate of car accidents can increase. Khazzoom (1994,1996) found no statistical relationship between vehicle weight and highway fatalities. If weight reductions are achieved via a switch to lighter material rather than through downsizing, there may not be any significant effect on fatalities. This debate is not closed as the fatality rate may also depend on the composition of the vehicle stock. The increased use of light trucks (that escape the CAFE regulations) may increase fatality rates for cars. In conclusion, air pollution reduction may be an ancillary benefit, negative effects on fatalities are probably small so that there is no compensating increase in ancillary costs. Finally the effects to be expected from suppressed traffic are small too.

Greene has studied the past performance of CAFE policies. It is not obvious that stronger CAFE standards are the best instrument to reduce GHG emissions in the future. Dowlatabadi, Lave and Russell (1996) conclude that CAFE regulations do indeed reduce GHG emissions but they are not a free lunch as they remain costly and do not necessarily reduce the urban ozone concentrations. They think that there may be cheaper ways to reduce CO<sub>2</sub> emissions than through fuel efficiency regulations in the transport sector.

The study of external costs of transport, the basic ingredient for estimating ancillary benefits has recently received more attention (see Greene, Jones, Delucchi,(1997)). Other transport and environment policies that have received attention in the last years are subsidies to alternative fuel vehicles (Kazimi, (1997)), accelerated scrapping schemes (Alberini *et al.* (1996) and pay at the pump insurance schemes (Kavalec and Woods,(1999)).

### 4.3 *Developing Countries*

There exist almost no systematic discussions of the economics of transport and environment in developing countries and GHG emission reduction in the transport sector<sup>63</sup>. It may be useful to line up differences and parallels with the OECD countries.

The policy discussion in developing countries will be different on three points.

First cars used in developing countries will not be as clean as in Europe or the USA: there exist many old vehicles and the technology used in new vehicles is not always the most recent one. This means that conventional air pollution emissions can be 5 to 10 times as high as in OECD countries and that reduction of conventional emissions can be an important source of ancillary benefits.

Second, although the same fuel tax and compulsory insurance policies can be used as in OECD countries, the monitoring and the enforcement of these policies are much weaker. For this reason, accident costs will be internalised to a smaller degree and savings in external accident costs can be an important source of ancillary benefits.

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<sup>63</sup> Except for work by the World Bank on CO<sub>2</sub> and transport [see Schipper and Marie-Lilliu (1999)] and work on air pollution damages presented at this symposium.

Third, transport needs and urbanisation are growing at a much higher pace in developing countries than in the OECD countries. This means that road expansion decisions, mass transit investments and land use are crucial and interdependent policy decisions. There are opportunities to realise efficiency gains and to reduce the volume of road traffic and emissions by integrating better road pricing policies and better mass transit policies. Land use policies are very important too but the contribution of economic knowledge is limited to a list of “errors not to make” rather than a full understanding of the optimal policy.

## **5. Conclusions**

The road transport sector is characterised by many important external costs so there is a potential for ancillary benefits of GHG reduction policies in this sector. The relative importance of the different externalities and their impact on the ranking of policies will be different.

In OECD countries there exist strong emission regulations and an enforced system of accidents insurance and liability rules. This explains why the most important external costs are congestion and to a much smaller extent accidents and air pollution. The traditional GHG reduction policies (high fuel taxes in the EU and strong fuel efficiency policies in the USA) have already been used intensively in the past. They are not very cost-effective and there are no important ancillary benefits to be expected from them. More interesting instruments are time and place differentiated pricing of transport that address congestion externalities directly and could generate a reduction of GHG emissions as by product. These policies need to be tailored to the local transport needs and require an integrated assessment area by area. The methodology for these studies exists but applications are still scarce.

In developing countries there are strongly growing transport needs and poorly enforced emission regulation, accident insurance and liability systems. Strongly growing transport needs imply that road expansion decisions, mass transit investments and land use planning are the major instruments. Poorly enforced accident insurance and emission regulations imply that external accident costs and external air pollution costs can be an important source of ancillary benefits. In order to use the same type of integrated assessment tools as in the OECD, these tools need to be extended to include better the land use policies and infrastructure extension. This remains an intellectual challenge.

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## **II. CASE STUDIES OF ANCILLARY COSTS AND BENEFITS**



## PRELIMINARY ESTIMATION OF THE POTENTIAL ANCILLARY BENEFITS FOR CHILE

by Luis A. CIFUENTES, Enzo SAUMA, Hector JORQUERA and Felipe SOTO

### 1. Introduction

There is no doubt that human activity is responsible for the increasing atmospheric concentrations of greenhouse gases (GHG), including carbon dioxide (CO<sub>2</sub>), methane (CH<sub>4</sub>) and nitrous oxide (N<sub>2</sub>O). The main activities responsible for this increase are fossil fuel combustion, which has grown at a rate unprecedented in human history, and changes in land use and agricultural practices. In the absence of emission controls for GHG, their atmospheric concentrations will rise in the next decades to levels that may induce changes in the climate of the earth. The Intergovernmental Panel on Climate Change (IPCC) estimates that human-induced climate change will increase surface temperatures by about 2°C by the year 2100 (Houghton, Meiro Filho *et al.* 1996), although many uncertainties exist about this estimate.

The climate change protocol signed in the Third Conference of the Parties in Kyoto in December 1997 set goals for emissions reduction for countries included in Annex I, which includes only developed countries. Non Annex I countries, mainly developing countries, do not need to abide to any emission reductions. The protocol set up an emissions trading framework that would allow countries (mainly Annex I) to invest in GHG reduction projects in other countries, and share part of the emissions credits. The implementation of such schemes, like “Joint Implementation” and “Clean Development Mechanisms” have been widely discussed at the subsequent Conference of the Parties held in Buenos Aires and Bonn.

In order to stabilize the global concentrations of GHG, it will be necessary for all countries, including developing countries, to make reductions in their emissions. However, developing countries shall make the most progress in reducing the growth of their greenhouse gas emissions by implementing measures that are consistent with their development objectives and that provide near term economic and environmental benefits. Within the existing framework, it is not clear for a developing country if it is beneficial to enter voluntarily in an emission reduction scheme. Our own analysis for Chile (Montero, Cifuentes *et al.* 2000) shows non-conclusive results, with the economic convenience depending heavily upon the initial emissions baseline assigned to the country.

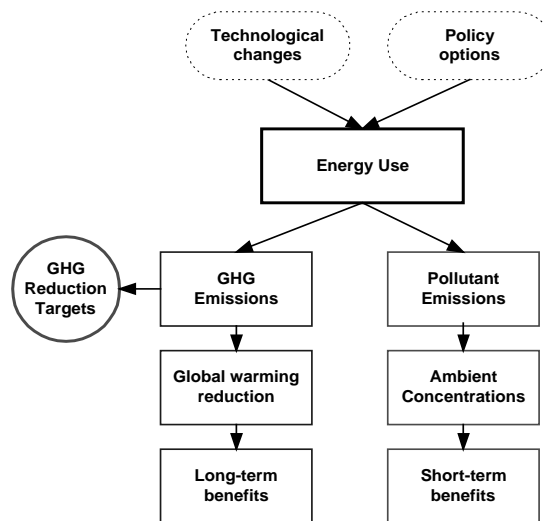
While many developing countries have conducted extensive analysis of possible greenhouse gas mitigation measures, relatively little attention has been given to full characterization of the more immediate environmental and health benefits that would result from these measures. Understanding those benefits has been a critical gap in past efforts to help developing countries estimate the cost of GHG abatement policies. Improving a country's understanding of the scope and potential magnitude of those direct public health benefits can help policy makers to take better decisions, by considering the full impact of adopting alternative climate change mitigation policies.

Figure 1 shows a schematic view of the potential social benefits resulting from measures aimed at reducing GHG emissions. Technological changes and policy options aim at reducing energy use to achieve the target in GHG emissions reductions. The path in the left side of the diagram shows that global warming reductions lead to long-term benefits, such as reduced extreme weather events, sea level rise, and communicable diseases spread, among others. However, these benefits are uncertain, at least to the same extent that global warming itself is uncertain. Also, from the standpoint of a single country, reducing the threat of global warming can be seen as a public good. Therefore, it is not strange that developing countries are more worried with local, immediate environmental and human health needs, such as control of air and water pollution, than with long-term problems such as global warming.

Nonetheless, the right path of the diagram in Figure 1 shows another set of benefits stemming from the measures aimed at reducing GHG emissions. In fact, the same combustion processes that lead to emissions of GHG also produce local and regional pollutants, like particulate matter (PM), sulfur dioxide (SO<sub>2</sub>), and nitrogen oxides (NO<sub>x</sub>). Thus, any measure aimed at reducing GHG emissions that also produces concomitant reductions in those pollutants, will lead to short and mid-term benefits from air pollution reduction, such as reductions in health effects associated to air pollution, reduction in vegetation and materials damages, and visibility improvements. Since these benefits can be considered a 'side effect' of the GHG mitigation measures, they are referred to as 'ancillary' benefits.

If properly assessed, consideration of these ancillary benefits may allow for implementation of policy measures that would otherwise have not been taken. If the ancillary benefits exceed the mitigation costs, they may even allow for "no regrets" GHG abating measures, in which taking immediate action to reduce GHG will be justified only by those benefits, even without consideration of the long-term benefits from GHG emissions reduction. Of all these benefits, those associated to health effects are probably the more important ones. They are the benefits considered in this report.

Figure 1. Short and long term social benefits derived from measures aimed at reducing GHG emissions

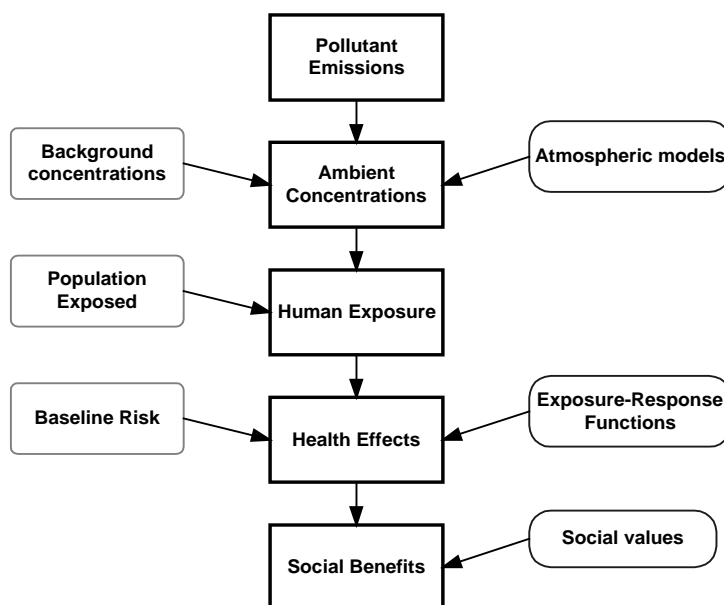


## 2. Methods

There are several levels at which the analysis of ancillary benefits of GHG mitigation can be conducted. The most detailed would be an analysis of individual mitigation measures, in which the changes in GHG and pollutant emissions associated to each policy or technological measure are estimated, and linked to a change in health effects in the population. This requires a great deal of data. The impacts of different mitigating measures are likely to vary according to the location and duration of the reduction in emissions, the population density close to the sources, and the prevailing meteorological conditions. In this work we took a global approach, conducting the analysis at an aggregate level for the whole country.

The first step to estimate the short-term health benefits is to link each policy or technological measure to the reduction in emissions pollutants. Once the changes in pollutant emissions have been assessed, it is necessary to link them to changes in ambient concentrations, population exposure, health effects and social benefits, using the Damage Function Method, showed schematically in the Figure 2.

Figure 2. **Damage Function framework used to estimate the social benefits of a reduction in emissions of primary air pollutants**



Fine particulate matter ( $PM_{2.5}$ ) was used as a sentinel pollutant to estimate the change in health effects. We choose to concentrate on  $PM_{2.5}$  because studies conducted in the U.S. (Schwartz, Dockery *et al.*, 1996) as well as our own studies in Santiago (Cifuentes, Vega *et al.*, 2000) have shown that the fine fraction of particulate matter is more strongly associated to health effects, especially mortality effects, than the coarse fraction of  $PM_{10}$ .

To estimate the potential health benefits for the whole country, we assembled a database of the current exposure of the Chilean urban population to particulate matter. Several studies led by the National Commission of the Environment (CONAMA) have measured particulate air pollution in cities that comprise almost half of the country's urban population.

The changes in ambient concentrations of particulate matter were estimated using two methods: one based on source apportionment of ambient fine particles concentrations; the other was based in statistical associations between atmospheric pollutants. Although both methods were developed using data specific for Santiago, we applied their results to the whole country. This assumes that the atmospheric processes for the rest of the country are similar to Santiago's, which is a crude assumption. Unfortunately, due to limited data, this was the only option available to us at this time.

With the projected ambient concentrations for each policy scenario, we computed the population exposure in each year. Based on data of a previous study in which we estimated the social losses due to particulate air pollution in Santiago, we estimated the health damages for the CP and the BAU scenario, obtaining the social benefits as the difference of the two. In the next sections we describe in detail the methods used in each step.

### 3. Emissions scenarios

We have considered two emissions scenarios: the Business-as-usual scenario (BAU), in which no GHG mitigation measures are taken, and a Climate Policy scenario (CP), in which measures are taken to reduce emissions of GHG.

We have relied on the results obtained in a previous study contracted by the Chilean Environmental Commission to the Research Program on Energy of the University of Chile (PRIEN 1999). The study projected the emissions for several greenhouse gases, including carbon dioxide (CO<sub>2</sub>), nitrous oxide (N<sub>2</sub>O) and methane (CH<sub>4</sub>) and several primary pollutants, including sulfur dioxide (SO<sub>2</sub>), nitrogen oxides (NO<sub>x</sub>) and non-methane hydrocarbons (NMHCs). Those projections were based on an engineering, bottom-up approach, considering technological measures like efficiency improvements and fuel switching to obtain emissions reductions. For the base case, policies that are currently in place and those which are scheduled to be applied were considered. In particular, all the measures of the Decontamination Plan for the Metropolitan Region that are scheduled to be implemented in Santiago were considered (Comisión Nacional del Medio Ambiente 1997), as well as the future investments in infrastructure contained in the national strategic plan developed by the Transportation Ministry (MOP 1997).

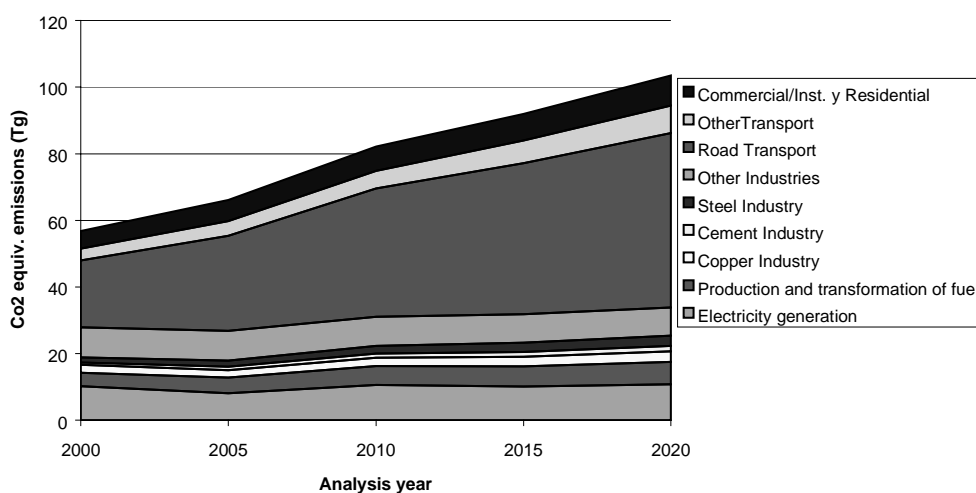
The most relevant assumptions considered for the projection of the base scenario from 2000 up to 2020 are:

- An average annual GDP growth of 4.5% for the whole period of analysis (2000 - 2020).
- A urban population increase of 1.9 % per year during the whole period.
- A constant rural population of around 2.5 million people.
- No substantial variations in the prices of energy.

#### 3.1 *Business as usual scenario*

The Business-as-usual scenario projected the emissions of CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O for the years 2000 to 2020, in 5-year intervals. Based on the Global Warming Potential (GWP) recommended by IPCC for each greenhouse gas (IPCC 1996) we computed the CO<sub>2</sub> equivalent emissions for each period. The emissions were projected for the different sub-sectors of the economy. For the analysis, we aggregated the data into the most relevant sub-sectors, as shown in the next figure.

Figure 3. CO<sub>2</sub>-equivalent emissions by sector for the BAU scenario



Source: Aggregation of data from (PRIEN 1999).

It can be observed in the figure that the baseline CO<sub>2</sub>-equivalent emissions would grow around 80% during the period 2000-2020 for the BAU scenario. This high growth is explained mainly by the explosive growth of the emissions in the road transport sector, as is clearly seen in the figure.

### 3.2 Emission reduction potential in the Climate Policy scenario

We consider as Climate Policy (CP) scenario the mitigation scenario developed by the Research Program in Energy of the University of Chile (PRIEN 1999). This mitigation scenario was developed following the bottom-up (or engineering) approach, considering the introduction of newer, more efficient technologies and computing the incremental cost and emissions reductions. Since the objective of PRIEN's study was to estimate emissions reductions that could be achieved through "no-regrets" implementation of technologies, the adoption and rate of penetration of the technologies was determined such that they would represent a net cost saving to the user. New technologies were considered for all sectors: residential, commercial, industrial and transport. Technologies considered in the residential/commercial sector included for example improved appliances and compact-fluorescent lamps. In the industrial sector, the main technologies considered were more efficient electric motors and the increased use of co-generation. In the transport sector the main mitigation measures were mode switching to cleaner means of transportation, and improvements in the fuel efficiency of the existing means of transport.

Due to the way the mitigation scenario was constructed, we can assume that mitigation costs are negative or close to zero. Therefore, the associated reductions in greenhouse gas emissions are relatively small. This may be a serious limitation, since we are then computing the ancillary benefits for the first mitigating measures in terms of control cost, without going up the marginal cost mitigation curve. If the mix of GHG and local pollutant emission reductions change for these measures, then the estimates for the ancillary benefits will also change.

The next table presents the projected CO<sub>2</sub> equivalent emissions reductions for the CP scenario, compared to the BAU scenario, for the years 2010 and 2020. The Steel Industry and the Other Industries sub-sectors show the biggest percentage reductions, of 23% and 20% respectively, for the year 2020. However, the biggest reduction in mass corresponds to the road transport subsector.

Table 1. CO<sub>2</sub>-eq emissions reductions by subsector of the economy (Tg)

Sector	Emissions (Tg)			Reductions CP with respect to BAU	
	2010	2010	2020	2010	2020
Electricity generation	10.2	10.6	10.8	6.2%	14.9%
Production and transformation of fuel	4.1	5.6	6.7	3.4%	8.4%
Copper Industry	2.4	2.5	3.2	4.9%	10.3%
Cement Industry	0.7	1.2	1.6	0.0%	0.0%
Steel Industry	1.4	2.3	3.2	16.1%	23.2%
Other Industries	9.1	8.8	8.4	9.8%	20.3%
Road Transport	20.1	38.4	52.4	6.2%	14.9%
OtherTransport	3.6	5.4	8.2	-0.6%	-0.9%
Commercial/Inst. y Residential	5.2	7.3	9.1	3.7%	8.1%
<b>Total</b>	<b>56.8</b>	<b>82.2</b>	<b>103.5</b>	<b>5.9%</b>	<b>12.9%</b>

Source: aggregation of data from (PRIEN 1999).

The next table presents the emissions reductions for the CP scenario, compared to the BAU scenario, for the GHG and the primary pollutants. The percentage reductions for all primary pollutants is similar to the reductions in CO<sub>2</sub>-equivalent, except for SO<sub>2</sub>, for which the reduction is slightly higher, due to the introduction of compressed natural gas in the country (starting at Santiago and other major cities) and to the sulfur reduction program in liquid fuels (gasoline and diesel).

Table 2. Emission reductions for each primary pollutant in 2010 and 2020

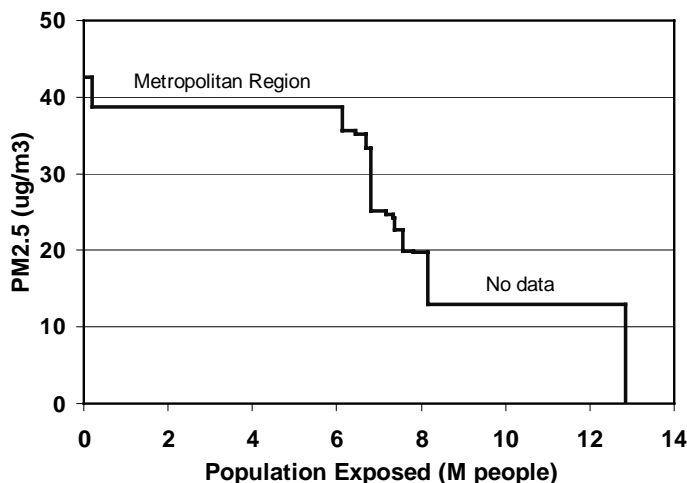
Pollutant	2010		2020	
	Gg	% red	Gg	% red
CO <sub>2</sub> eq	4,833	5.9	13,392	12.9
CO <sub>2</sub>	4,537	6.0	12,404	13.1
CH <sub>4</sub>	2.95	2.4	7.12	5.4
N <sub>2</sub> O	0.75	5.5	2.71	14.4
CO	33.14	4.0	102.30	11.2
SO <sub>2</sub>	7.15	6.7	15.88	14.9
NO <sub>x</sub>	15.92	5.2	40.87	10.7
VOCNM	5.10	4.3	15.52	11.6

#### 4. Human exposure to air pollution in Chile

Chile has a widespread ambient particulate matter pollution problem. Santiago, the capital of Chile, is one of the world's most polluted cities by particulate matter. Regular daily measurements of  $PM_{2.5}$  and  $PM_{10}$  using dichotomous samplers began in 1988 in five stations across the city. The original network was expanded in 1997 with a new network of eight monitoring stations. Regular monitoring is not currently conducted in any other city, except a few localities close to megasources like copper smelters and power plants, where the law mandates regular monitoring to ensure that ambient air quality levels are not violated. Some research projects [(SESMA 1999), (CIMM 1998), (Gredis 1999), (Cosude 1999)] have conducted sporadic measurements in several other cities though.

We gathered all the available concentrations data to estimate the current level of exposure of the Chilean population to fine particulate matter. All cities which have some particulate matter measurement, either  $PM_{2.5}$  or  $PM_{10}$ , comprise a total of 7.8 million people, or about 63% of the urban population of Chile in 2000. For cities that did not have measurements of  $PM_{2.5}$ , we estimated them from the  $PM_{10}$  concentrations, based on the national average ratio of  $PM_{2.5}$  to  $PM_{10}$ . For those cities with no measurements at all, we assumed a level equal to the cleanest city measured, that is,  $13 \mu\text{g}/\text{m}^3$ . This assumption is probably an underestimate, since some industrial zones that currently lack measurements, probably have a higher concentration. The next figure shows the estimated exposure to fine particulate matter for all the urban population in Chile. The figure underlines the relative importance of the Metropolitan Region of Santiago in the total exposure of Chilean population. The total population exposure in 2000 will be 344.5 million person\* ( $\mu\text{g}/\text{m}^3$ ) of  $PM_{2.5}$ , of which 66% correspond to the Metropolitan Region.

Figure 4. Population exposure of the Chilean urban population to  $PM_{2.5}$  in 2000





## 5. Changes in air pollutant concentrations due to changes in emissions of primary pollutants

This step is a crucial part of the method linking emissions of primary pollutant to social losses. For a detailed analysis, it should rely on atmospheric dispersion models, specifically in models that incorporate the complex set of chemical reactions occurring in the atmosphere. None of those models is available for Chile at this time. For this analysis, we estimated the impacts of emissions changes on PM concentrations based on two approximate methods, described in the following sections.

### 5.1 Method 1: Use of a box model to develop emission concentration relationships

A simplified methodology was used to estimate the future impacts of PM<sub>10</sub>, PM<sub>2.5</sub> and coarse fractions. The starting point is the Eulerian Box model approach that reads:

$$\frac{dC_i}{dt} = \frac{q_i}{H(t)} + R_i - \frac{V_{d,i}}{H(t)} C_i + \frac{u(t)}{\Delta x} (C_i^U - C_i) + \frac{(C_i^A - C_i)}{H(t)} \frac{dH}{dt} \quad (1)$$

where  $C_i$  is the pollutant concentration ( $i = CO, SO_2, PM_{10}, PM_{2.5}$ , etc.),  $H(t)$  is the mixing height,  $q_i$  the surface emission within the box,  $R_i$  the net production rate by chemical mechanisms,  $V_{d,i}$  the deposition flux (dry and wet) at the ground surface,  $u(t)$  the average wind speed in the box and the superscripts  $U$  and  $A$  stand for upwind and aloft advected concentrations, respectively. The rightmost term on the right hand side of (1) is only applicable whenever the mixing height is rising, that is, from sunrise until early afternoon (Seinfeld and Pandis 1998).

The above equation describes mathematically the concentration of species above a given area, accounting for emissions, chemical reactions, removal, advection of material in and out of the airshed and entrainment of material during growth of the mixed layer. The strongest assumption is that the corresponding airshed is well mixed.

If equation (1) is integrated for a pollutant like CO or SO<sub>2</sub>, and assuming a first order decay process, it can be shown that the following relationship holds:

$$\langle C_i \rangle = \left\langle \frac{q_i}{k_i \cdot H + v_{d,i} + \frac{u \cdot H}{\Delta x}} \right\rangle + \left\langle \frac{C_i^o}{1 + \frac{k_i \cdot \Delta x}{u} + \frac{v_{d,i} \cdot \Delta x}{H \cdot u}} \right\rangle + \delta_i \quad (2)$$

The above equation is a linear relationship between emissions and concentrations, and it was used to generate long-term forecasts of CO and SO<sub>2</sub> for Santiago for 2000-2020. The emissions of CO and SO<sub>2</sub> come from fuel consumption, so the model parameters were calibrated using measured ambient concentrations and historical data on fuel consumption and fuel sulfur content.

Next, in order to model the emission term for particulate matter fractions, it was assumed that the emissions of particulate matter can be expressed as a sum of contributions coming from mobile sources, stationary sources and other sources in the following manner:

$$q_{PM10} = \alpha(q_{CO})_{Mobile\ Sources} + \beta(q_{SO2})_{Stationary\ Sources} + \gamma \quad (3)$$

where

$\alpha$  = ratio of  $PM_{10}$  to CO emissions in the mobile sources

$\beta$  = ratio of  $PM_{10}$  to  $SO_2$  emissions in the stationary (industrial, commercial and residential) sources

$\gamma$  = emissions not directly linked to mobile or stationary source emissions

Therefore,  $\alpha$  stands for the ratio of  $PM_{10}/CO$  in the emissions from the fleet in Santiago,  $\beta$  represents the ratio of  $PM_{10}/SO_2$  emissions in industrial and residential sources and  $\gamma$  is a term independent of those emissions, and it is associated to mechanisms such as construction activities, wind erosion, agricultural activities, forest fires, etc.

Using equation (2) for CO and  $SO_2$  and inserting equation (3) within the box model equation for  $PM_{10}$  leads, after some manipulation and simplification to:

$$\langle C_{PM10} \rangle = a \langle C_{CO} \rangle + b \langle C_{SO2} \rangle + \frac{c}{u} + d \frac{P}{u} + e \quad (4)$$

Where  $u$  is the average wind speed and  $P$  the total precipitation recorded (this takes into account of the wet deposition term). Therefore, a linear regression for the daily averages of  $PM_{10}$ ,  $PM_{2.5}$  and coarse fractions against the daily averages of CO,  $SO_2$ ,  $(1/u)$  and  $(P/u)$  will produce estimates of the unknown parameters in the model.

### 5.1.1 Parameter estimation and model validation

In order to validate the above model, data gathered at Santiago for the fall and winter seasons from 1990 to 1994 were used to fit the model (in some cases, data from 1995 and 1996 were used to increase the database, as was the case in Station C). The air quality data came from the MACAM monitoring network, and included hourly measurements of CO,  $SO_2$ , and surface wind speed  $u$  plus daily measurements of  $PM_{10}$ ,  $PM_{2.5}$  and coarse fractions. A substantial amount of time was devoted to extracting daily averages of the different terms appearing in equation 2, considering missing values, analyzing partial scatter plots to detect outliers, and so on. Model parameters were obtained by using classical, linear regression analysis of equation (4). In this fashion, we could estimate the model parameters for stations A, B, C and D of the MACAM network, and for the three fractions:  $PM_{10}$ ,  $PM_{2.5}$  and coarse particles. In particular, we can estimate the different contributions to the total, ambient particle concentrations coming from:

- a) Advected and secondary particles, lumped together in the  $c/u$  term in equation (4)
- b) Directly emitted by mobile sources, and so proportional to CO concentrations
- c) Directly emitted by stationary sources, and so proportional to SO<sub>2</sub> concentrations
- d) Deposited onto the ground by wet precipitation removal

From the 1997 Emission Inventory for Santiago (EIS), as developed by CENMA (1997), the estimated ratio of PM<sub>10</sub> emissions from mobile sources to total CO emissions is:

$$\frac{2730(\text{ton} / \text{yr})}{244921(\text{ton} / \text{yr})} = 0.011(\text{g} / \text{g})$$

The  $a$  coefficients for the CO concentration in the PM<sub>10</sub> model have the values 10.25, 9.97, 19.57 and 7.78 at stations A, B, C and D, respectively, when CO is measured in ppm and PM<sub>10</sub> in (µg/m<sup>3</sup>). In units of (g/g), the coefficients take the values 0.009, 0.0087, 0.017 and 0.0068 for stations A, B, C and D, respectively. All coefficients are significant ( $p < 0.05$ ). The similar results among monitoring sites and their reasonable agreement with the value estimated above from the annual emission inventory for Santiago show that the box model is capable of reflecting these relationships among primary emissions.

From the 1997 EIS the ratio of PM<sub>10</sub> to SO<sub>2</sub> emissions for the stationary sources is:

$$\frac{3175 / \text{ton} / \text{yr}}{21169(\text{ton} / \text{yr})} = 0.15(\text{g} / \text{g})$$

In this same units, the fitted values for  $b$  are 0.24, 0.21, 0.50 and 0.58, with all of them being significant ( $p < 0.05$ ) for stations A, B, C and D, respectively. The reason for  $b$  values higher than the value given by the emission inventory is that the ratio of PM to SO<sub>2</sub> is enhanced by the faster removal of SO<sub>2</sub> from the gas phase. In other words, by the time emissions reach a monitor site, a significant amount of SO<sub>2</sub> has already been deposited or degraded by chemical or physical mechanisms. This is more evident for stations C and D, which are rather away from major traffic lanes and so tend to be impacted by rather aged plumes, associated with regional scale dispersion of sulfur in Central Chile. On the other hand, stations A and B are located near busy streets, so they are impacted by fresh emissions coming from mobile sources. By contrast, this effect does not show up for CO, because its rate of chemical oxidation is fairly low, and so is its deposition velocity (Seinfeld and Pandis, 1998).

In addition, the intercepts ( $e$  coefficients) on the lineal regression equation produce estimates of the background levels of PM<sub>10</sub>, PM<sub>2.5</sub> and coarse fractions. This is relevant information to be used in the estimation of future concentration impacts. We have estimated that background levels of PM<sub>10</sub>, PM<sub>2.5</sub> and coarse particles are around 45, 27 and 18 (µg/m<sup>3</sup>), respectively. These three values compare very well with the measurements made by (Artaxo 1998) at Buin, a rural site 35 km south of Santiago considered representative of upwind, background values for the greater Santiago area. The values reported by Artaxo *et al.* in the winter 1996 campaign were 52, 29 and 23 (µg/m<sup>3</sup>), for PM<sub>10</sub>, fine and coarse particles respectively. The major difference lies in the coarse fraction, but (Artaxo 1998) measured PM<sub>2.0</sub> as fine fraction, thus explaining their larger estimates of the coarse particle background.

We have to recall that the box model cannot account for the generation of secondary aerosols (mostly sulfates and nitrates), because the chemistry of these processes is far too complex to be included within a simplified model like this one. We cannot estimate the magnitude of this uncertainty until a comprehensive simulation of those processes is carried out for Santiago. Nevertheless, the model parameters were fitted using actual data recorded at the monitoring network, so the model should represent reliably the PM levels within the city.

### 5.1.2 Projection of future impacts

From the previous results, the working equation to estimate future concentrations under new emission scenarios is obtained from (4) in the following way

$$\langle C_{PM10} \rangle_{2XYZ} = a \langle C_{CO} \rangle_{2XYZ} + b \langle C_{SO2} \rangle_{2XYZ} + \left( \frac{c}{V} \right)_{HISTORICAL} + d \left( \frac{P}{V} \right)_{HISTORICAL} + e_{HISTORICAL} \quad (5)$$

Where 2XYZ stands for any future scenario. In addition:

- a) The CO and SO<sub>2</sub> concentrations are forecasted using the box models calibrated with historical data from 1990 to 1998.
- b) It will be assumed that Santiago will follow the same trend in emissions as the whole country in the PRIEN annual emission forecasts.
- c) The estimates of contributions of resuspended dust and wet scavenged particle concentrations will be assumed to stay in the same values as in the model calibration period. That is, we assume that the emission factor for resuspended particles will stay the same. Given the uncertainties in estimating this type of emission factor, we consider the above approximation reasonable; for instance, (Venkatram 1999) have reported estimates for this emission factor between 0.1 and 10 g/VKT for a metropolitan area (VKT are the total kilometers traveled by all vehicles in a given period).
- d) The proportion of particles that are deposited by wet mechanisms is assumed to be the same as the values computed from the regression analyses: about 1 to 2% for most of the fractions. This means that, at least for Santiago, these quantities can also be incorporated in equation (5) as fixed proportions of the total, average concentration  $\langle C_i \rangle$  therein.

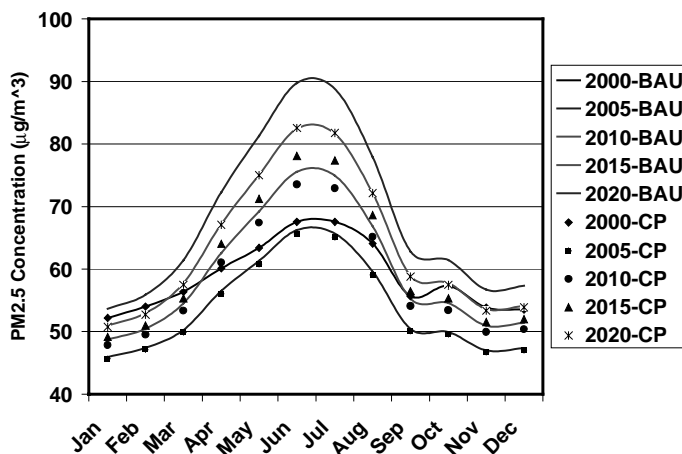
### 5.1.3 Results of the simulated scenarios

In order to simulate impacts for the BAU and CP scenarios, the following specific assumptions were made:

- i) Background concentrations were kept at the same values as 1994. Although (Artaxo 1998) have estimated long range contributions from copper smelters that will undergo emission reduction plans, these plans will be pursued regardless of the long-term GHG policies (if any) in the country, that is either under BAU or CP scenarios.
- ii) The parameters obtained for the different monitor stations will be kept fixed at their estimated values for the calibration period (1990-1996).

The next figure shows the projected impacts of PM<sub>2.5</sub> at monitoring station B; similar results hold for the other stations, so they are not shown here. It is clear that by 2020 the two scenarios achieve different impacts, with CP concentrations being lower by up to 7 µg/m<sup>3</sup>.

Figure 5. Projections of PM<sub>2.5</sub> concentrations at monitoring Station B



### 5.2 Method 2 : Source apportionment of fine particulate matter concentrations

In this approach, we estimated the changes in ambient PM concentrations due to changes in primary pollutant emissions using an alternative method. The method is based on source apportionment data of PM<sub>2.5</sub> concentrations to primary pollutants conducted in Santiago in 1996 and 1998 (Artaxo 1996; Artaxo 1998; Artaxo, Oyola *et al.* 1999). We computed the fraction of PM<sub>2.5</sub> concentrations in Santiago attributable to each primary pollutant, based on those measurements, and obtained the fractions shown in the next table.

Table 3. Percentage of PM<sub>2.5</sub> concentrations attributable to each primary pollutant in Santiago, 1998

Primary Pollutant	Percentage attributable	90% CI
Resuspended Dust	5.0%	(0.5% - 10%)
SO <sub>2</sub>	20.0%	(15.5% - 25%)
NMHC	0.0%	(0% - 0%)
NO <sub>x</sub>	30.0%	(21.1% - 39%)
PM <sub>10</sub>	33.5%	(24.6% - 42%)
Other	11.5%	

Source: own estimates based on (Artaxo 1996) , (Artaxo 1998) and (Artaxo, Oyola *et al.* 1999).

In the above table  $PM_{10}$  should be understood as primary emission of PM, whereas  $SO_2$  and  $NO_x$  are associated to secondary sulphates and nitrates, respectively. Assuming that the contribution of each primary pollutant remains fixed over time in the value given in Table 3 above, then the relative change in ambient  $PM_{2.5}$  concentrations can be expressed as:

$$\Delta\%[PM_{2.5}] = \sum_i F_i \cdot \Delta\%[P_i] \quad (6)$$

where

- $\Delta\%[PM_{2.5}]$  is the relative change in  $PM_{2.5}$  concentrations.
- $\Delta\%[P_i]$  is the relative change in pollutant i concentrations.
- $F_i$  is the fraction of  $PM_{2.5}$  apportioned to pollutant i, according to Table 3.

This equation should be applied only to the fraction of the  $PM_{2.5}$  concentrations above background concentrations. However, we should consider only the natural background, not the background due to emissions occurring elsewhere in the country. In effect, if we are conducting an analysis for the whole country, assuming a relatively uniform distribution of pollutant sources within the country, the background concentration in any given city will also change when the level of emissions changes within the whole country.

## 6. Health impact estimates

There is a growing number of studies linking particulate air pollution with both mortality and morbidity all over the world. For short term effects, the work of Dockery and Schwartz in the late eighties has been replicated in more than 40 cities to date (and the number keeps growing), although still most of the studies come from US and European cities. For chronic effects, two prospective studies conducted in the US, the Harvard Six cities study (Dockery, Pope III *et al.* 1993) and the Pope and colleagues study (Pope III, Thun *et al.* 1995) have shown significant results, in agreement with results from earlier cross-sectional studies (Lave and Seskin 1977). Although the causal mechanism by which exposure to particulate matter can induce death is not yet know, there is not much doubt than the association is not a spurious one, and the US has moved towards more stringent standards based on the recent studies (EPA 1997).

For morbidity effects, studies in several countries have associated particulate matter with a number of health endpoints, including hospital admissions, emergency room visits, increased incidence asthma attacks, work loss days, restricted activity days, and minor symptoms, as well as increased incidence of chronic bronchitis (EPA 1996).

Most of the studies linking air pollution and health are based on a Poisson model. In this model, the relative risk (RR) associated with a change in the PM concentrations is given by:

$$RR(\Delta PM) = \exp[\beta * \Delta PM] \quad (7)$$

The slope coefficient,  $\beta$ , is obtained from the epidemiological studies, as will be shown later.  $\Delta PM$  is the change in PM concentrations from a reference concentration. The relative risk needs to be applied to a base number of effects, which is obtained from the observed number of effects on the population that is exposed to a given level of air pollution. Therefore, the number of health effects at a given concentration  $C$ , is given by:

$$\text{Effects}(C) = \exp(\beta \cdot (C - C_0)) \cdot R_0 \cdot \text{Pop} \quad (8)$$

where  $R_0$  refers to the base rate of effects at concentration  $C_0$ , and is generally obtained from health statistics data, and  $\text{Pop}$  is the exposed population. The above formula assumes that there is no threshold in the effects. If there is a threshold in the effects, i.e. a concentration  $C_T$  below which there are no effects, the formula becomes:

$$\text{Effects}(C) = \exp(\beta \cdot (C - \max\{C_0, C_T\})) \cdot R_0 \cdot \text{Pop} \quad (9)$$

For some studies the above formula applies to daily effects, and the effects rate should be expressed as the number of effects per day. To obtain the number of excess effects in a year, it is necessary to add up the effects for all days of the year. If there is a threshold, the summation becomes more complicated. For computing the exact number of effects in this case it is necessary to know the form of the frequency distribution of the daily concentrations. Generally, it is assumed that daily concentrations follow a lognormal distribution (Ott 1990), although other distributions have been shown to better represent the physical process underlying air pollution concentrations (Morel *et al.*, 1999).

**Exposure-response functions.** We conducted the analysis based on exposure-response functions obtained from the literature, mainly from the estimation of benefits of the Clean Air Act performed by EPA (EPA 1997, EPA 1999) and from the recommendations of the World Health Organization by Ostro (Ostro 1996). We complemented these sources with exposure response functions from studies performed in Santiago. For mortality we used our own results (Cifuentes, Vega *et al.* 2000). For child medical visits, we used (Ostro, Eskeland *et al.* 1999) and (Illabaca, Olaeta *et al.* 1999). All of the studies correspond to short-term effects, except for chronic bronchitis and long-term exposure mortality. Following Ostro 1996, for mortality due to long-term exposure, we used the coefficient from the study of Pope *et al.* (Pope III, Thun *et al.* 1995) only for the high case, i.e., our mid estimate of mortality does not consider the chronic effects of pollution. Whenever possible, we used exposure-response functions based on  $PM_{2.5}$ . If they were available only for  $PM_{10}$ , we convert them to  $PM_{2.5}$  using the relation  $PM_{2.5} = 0.55 PM_{10}$ .

We considered three age groups in the analysis: Children 0-18 yrs, Adults, 18-64 yrs, and 65+ yrs, In some cases, we considered specific age groups, like for asthma attacks, in which the exposure-response functions are for children below 15 yrs. The summary of the exposure-response coefficients for the effects considered is shown in the next table.

Table 4. Summary of exposure-response coefficients used in the analysis

Endpoints	Age Group	$\beta$	$\sigma_{\beta}$	Source
Mortality (long term exp)	>30 yrs	0.00640	0.00151	Pope et al,1995
Chronic Bronchitis	> 30 yrs	0.02236	0.007891	Schwartz,1993
Mortality (short term exp.)	All	0.00120	0.000304	Cifuentes et al, 2000
Hospital Admissions RSP	> 65 yrs	0.00169	0.000447	Pooled
Hospital Admissions COPD	> 65 yrs	0.00257	0.000401	Pooled
Hosp. Adm Congestive heart failure	> 65 yrs	0.00135	0.000565	Schwartz & Morris, 1995
Hosp Adm Ischemic heart disease	> 65 yrs	0.00090	0.000400	Schwartz & Morris, 1995
Hospital Admissions Pneumonia	> 65 yrs	0.00134	0.000264	Pooled
Asthma Attacks	All	0.00144	0.000315	Ostro et al, 1991
Acute Bronchitis	8-12 yrs	0.00440	0.002160	Dockery et al., 1989
Child Medical Visits LRS	< 18 yrs	0.00083	0.000330	Ostro et al, 1999
Emergency Room Visits	All	0.00222	0.000427	Sunyer et al, 1993
Shortness of Breath (days)	< 18 yrs	0.00841	0.003630	Ostro et al, 1995
Work loss days (WLD)	18-65 yrs	0.00464	0.000352	Ostro et al, 1987
Restricted Act. Days (RAD)	18-65 yrs	0.00475	0.000288	Ostro et al, 1987
Minor Restricted Act. Days (MRAD)	18-65 yrs	0.00741	0.000704	Ostro et al, 1989

**Base rate of effects.** The other parameters needed to compute the total number of effects are the exposed population and the effects base rate. We projected the exposed population using the estimates of the Chilean Institute of Statistics, considering that the age distribution remains constant. For the base rate of the effects we used the rates for Santiago for all the cities.

## 7. Effects valuation

To estimate the social benefits associated to reduced health effects, it is necessary to estimate society's losses due to the occurrence of one extra effect. Several methods exist to value such losses. The most straightforward one is based on the direct losses to society stemming from the cost of treatment of each effect plus the productivity lost. This approach, known as the human capital method for mortality effects, and the cost of illness for morbidity effects, suffers from a serious limitation, by not considering the willingness to pay of the individuals to avoid the occurrence of an extra effect, or to reduce her risk of death. However, because values are easier to compute and defend, it has been used in previous analysis of quantification of air pollution effects, such as the economic valuation of the benefits associated to the Decontamination Plan of Santiago (Comisión Nacional del Medio Ambiente 1997).

We choose to use values that reflect the willingness to pay of individuals to reduce the occurrence of one extra effect. Since there are no such values available for Chile, the unit values of the effects are based on those used by the US EPA (EPA 1999), transferred to Chile using the ratio of the per capita income of both countries. By far, the more important effects are premature mortality. For these effects, we choose a lower bound from the range of values used by EPA, which became US\$338 thousand after adjustment, for the year 1997. This value falls within the range of values that we have obtained in a pilot test of a contingent valuation study of willingness to pay for reducing mortality risks in Santiago (Cifuentes, Prieto *et al.* 1999). For The summary of values used in the analysis in shown in the next table. The values were updated annually using a projected growth in real per capita income of 2.6%.



Table 5. **Unit values for each effect for the year 1997 (1997US\$ per effect)**

<b>Endpoint</b>	<b>mid</b>	<b>90% CI</b>
Mortality (long term exp)	281,209	(111,956 - 707,906)
Chronic Bronchitis	45,556	(22,192 - 68,921)
Mortality (short term exp.)	338,549	(134,785 - 852,252)
Hospital Admissions RSP	2,796	(2,796 - 2,796)
Hospital Admissions COPD	3,624	(3,597 - 3,651)
Hosp. Adm Congestive heart failure	3,832	(3,815 - 3,849)
Hosp. Adm Ischemic heart disease	4,755	(4,742 - 4,767)
Hospital Admissions Pneumonia	3,670	(3,654 - 3,686)
Asthma Attacks	7	(3 - 11)
Acute Bronchitis	10	(4 - 16)
Emergency Room Visits	54	(33 - 74)
Child Medical Visits	165	(133 - 198)
Shortness of Breath (days)	1	(0 - 2)
Work loss days (WLDs)	18	(18 - 18)
RADs	9	(5 - 12)
MRADs	8	(5 - 12)

Source: Values from EPA (1999) transferred for Chile using the ratio of per capita income.

## 8. Uncertainty and variability analysis

As has been shown in the preceding sections, each step of the analysis is fraught with uncertainty. Explicit consideration of all the uncertainties is crucial to illuminate the analysis for several reasons (Morgan and Henrion 1990):

- It lets us identify the important factors in the analysis.
- It can help us identify which steps of the analysis need to be improved the most.
- It points out potential sources of disagreement between different experts or analysts.

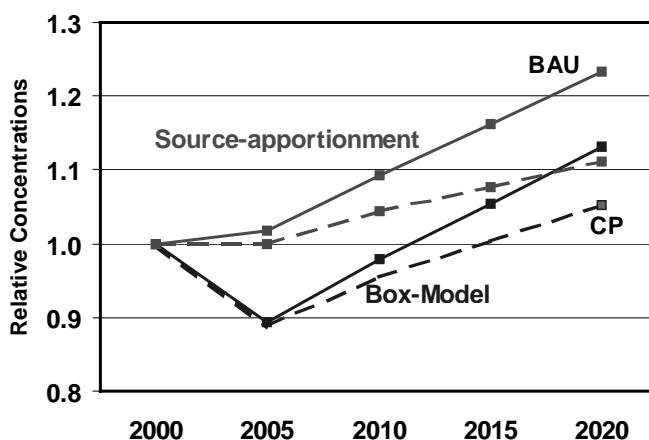
Uncertainty can be classified into parameter uncertainty, model uncertainty, and scenario uncertainty. In this analysis, we have considered explicitly only the first two. Parameter uncertainty can be modeled quantitatively treating the parameters as random variables. We have done so for the exposure-response coefficients for health effects quantification, for some parameters of the ambient concentration models, and for the unit values of the effects. A more difficult kind of uncertainty is model uncertainty. As discussed in Section 5, we have considered two different models to estimate the change in PM<sub>2.5</sub> concentrations due to changes in emissions, this was considered necessary because of the relevance of air pollution dispersion modeling in the framework depicted in Figure 2.

To consider quantitatively the uncertainty in the analysis, the model was implemented in the Analytica modeling environment (Lumina Decision Systems 1998), which is based on Montecarlo simulation. This very flexible modeling environment let us propagate and analyze the uncertainty of the parameters and the results.

## 9. Results

Based on the emissions changes presented in Section 3, we estimated the evolution of  $PM_{2.5}$  concentrations in time for both methods proposed in Section 5. The next Figure shows the mid estimates of the projected  $PM_{2.5}$  concentrations for each scenario, using both methods of estimating the concentrations. The concentrations are referred to the concentrations in the year 2000.

Figure 6. **PM2.5 concentrations relative to year 2000 concentrations, for both methods of estimating the concentrations**



The figure shows that both methods produce similar results for each scenario, BAU and CP, with the concentrations increase being driven mainly by the increase in  $NO_x$  and PM emissions. However, given the consideration of different primary pollutant emission changes, the difference between the BAU and CP scenarios is approximately 50% bigger for the source apportionment method. The more pronounced minimum in 2005 for the Box model approach is caused by the heavier weight given to CO and  $SO_2$  concentrations, with respect to the source apportionment approach.

Applying the changes in  $PM_{2.5}$  concentrations to the exposed population in each city it is possible to compute the excess health effects for each scenario. The next table shows the avoided excess health effects in the year 2010 and 2020. The excess effects have been computed assuming there is no threshold in any of the effects. The table shows the mid value of the effects for each policy scenario, grouped by type of effect, summed up over all age groups, and the 90% confidence interval. We show the results for the source apportionment method. The values for the Box model are smaller.

Table 6. **Avoided health effects for the years 2010 and 2020**

Endpoint	2010		2020	
	mid	90% CI	mid	90% CI
Premature Deaths	100	(62 - 431)	305	(189 - 1,290)
Chronic Bronchitis	710	(503 - 854)	2,157	(1,526 - 2,572)
Hospital Admissions	619	(480 - 797)	1,887	(1,450 - 2,423)
Emergency Room Visits	9,972	(6,431 - 14,882)	30,095	(19,654 - 44,984)
Child Medical Visits	4,837	(1,919 - 8,178)	14,642	(5,866 - 24,878)
Asthma Attacks & Bronchitis	133,022	(86,530 - 183,840)	399,351	(263,016 - 556,863)
Restricted Activity Days	2,878,743	(1,868,859 - 3,716,428)	8,804,442	(5,660,315 - 11,270,793)

Note: PM<sub>2.5</sub> concentration changes estimated using source apportionment method, equation (6).

The next table shows the total number of effects avoided from 2000 to 2020 for the BAU-CP scenario comparison.

Table 7. **Total number of health effects avoided in the CP scenario with respect to the BAU scenario during the period 2000 to 2020**

Endpoint	Total effects avoided	
	mid	90% CI
Premature Deaths	2,771	(1,546 - 10,840)
Chronic Bronchitis	18,130	(10,710 - 22,170)
Hospital Admissions	15,000	(12,930 - 20,760)
Emergency Room Visits	247,200	(166,600 - 353,400)
Child Medical Visits	118,600	(47,560 - 205,400)
Asthma Attacks & Bronchitis	3,339,000	(1,981,000 - 4,998,000)
Restricted Activity Days	75,430,000	(43,650,000 - 96,670,000)

Note: PM<sub>2.5</sub> concentration changes estimated using source apportionment method, equation (6).

For the whole period of analysis, the mid estimate is around 2,800 deaths that can be avoided, with a 90% confidence interval of 1,500 to 10,800 (the upper bound of this interval is high because it includes long-term exposure deaths). Most of these effects will occur in the Metropolitan Region of Santiago.

Using the unit values shown in the preceding chapter, we computed society's social losses due to these health effects. The difference of the damages for each scenario is the social benefit of the mitigation measures.

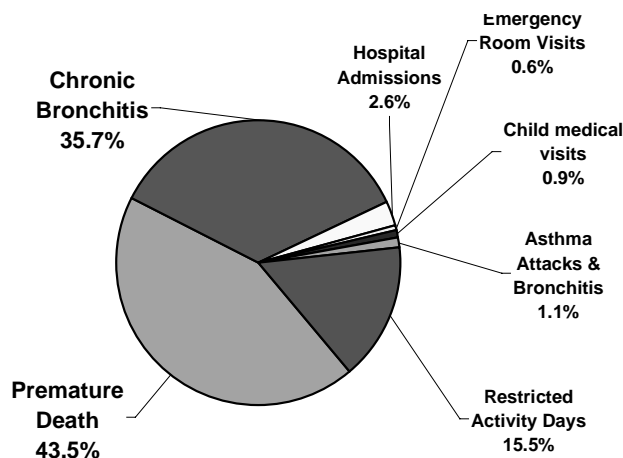
Table 8. Social benefits for 2010 and 2020 (Millions of 1997US\$)

Endpoint	2010		2020	
	mid	90% CI	mid	90% CI
Premature Deaths	53.0	(15.1 - 371.3)	210.6	(60.3 - 1,494.0)
Chronic Bronchitis	41.8	(26.8 - 67.3)	168.4	(106.8 - 265.8)
Hospital Admissions	3.2	(2.6 - 3.9)	12.8	(10.4 - 15.7)
Emergency Room Visits	0.7	(0.4 - 1.1)	2.9	(1.6 - 4.6)
Child Medical Visits	1.1	(0.5 - 2.0)	4.3	(1.9 - 7.9)
Asthma Attacks & Bronchitis	1.3	(0.5 - 2.4)	5.3	(2.2 - 9.5)
Restricted Activity Days	18.4	(14.4 - 23.9)	74.0	(56.4 - 94.9)
<b>Total</b>	<b>119.6</b>		<b>478.2</b>	

Note: PM<sub>2.5</sub> concentration changes estimated using source apportionment method, equation (6).

Where do these benefits come from? The next figure shows the share of the social benefits for each effect. It is clear that the biggest share of the benefits comes from avoided premature mortality, although chronic bronchitis cases also have an important contribution in the mid value case. Premature mortality dominates the values for the upper bound of the confidence interval, representing around 70% of the benefits, mainly due to the consideration of long-term exposure deaths estimates in that case.

Figure 7. Share of the present value of benefits for each type of effect (mid estimates)



Note: Based on mid estimates using source apportionment method.

All the previous results have been obtained the using source apportionment model to estimate the change in PM<sub>2.5</sub> concentrations. The next table shows the net present value of the benefits, computed using a real discount rate of 12% (the rate used in Chile for evaluation of all social projects) for the two models for computing the changes in PM<sub>2.5</sub> concentrations.

Table 9. **Present value of social benefits for each method of emissions impacts estimation**  
(Million of 1997US\$)

Method for estimating PM2.5 concentrations	90% CI		
	mid		
Source apportionment	710	314	2,472
Box Model	417	194	1,376

Finally, another way to look at these results is to compute the average social benefit accrued from the reduction of each ton of carbon. This is obtained by simply dividing the benefits by the equivalent carbon reductions in each year.

Table 10. **Average social benefit per ton of carbon (1997US\$/tonC)**

Year	Atmospheric Model		
	Source appmt	Box Model	Avg of two models
2010	90	48	69
	(42 - 337)	(21 - 190)	(21 - 337)
2020	129	79	104
	(60 - 479)	(39 - 284)	(39 - 479)

## 10. Discussion

This work is a preliminary estimation of the potential ancillary benefits of greenhouse mitigation in Chile. We have conducted an aggregate analysis for the whole country, based on previously developed base (BAU) and mitigation (CP) scenarios.

The results show potentially high ancillary social benefits. The implementation of the CP scenario may prevent 2,800 deaths in the period 2000 to 2020, with a range from 1,500 up to 10,800. The mid estimate rests on generally accepted concentration-response coefficients, and which are in agreement with studies conducted in Santiago, Chile's capital, which accounts for most of the exposure to particulate matter. The upper bound of the confidence interval relies heavily on the mortality estimates from prospective studies performed in the U.S., under different conditions than in Chile, so their application is more uncertain.

From an economic standpoint, the potential ancillary benefits represent a substantial fraction of the potential costs of the mitigating options. For 2010, the benefits per ton of carbon abated range from 21 up to 337 dollars, depending on the models used to estimate the impact of emissions on concentrations. For 2020, the values range from 39 to 479 dollars per ton of carbon abated. The magnitude of these values is comparable to current estimates of abatement costs of carbon, for modest mitigation scenarios. Therefore, these ancillary benefits may offset a significant fraction of the costs needed to implement the measures. In the specific case studied here, in which all the measures considered do not impose a cost on the user, these ancillary benefits indicate a net benefit for society.

However, it is necessary to stress the limitations of the analysis. The main one is that it has been conducted at an aggregate level for the whole country, with no consideration of local conditions, like emissions, meteorology or population density surrounding the sources. Therefore, our estimates are average estimates across all these dimensions. Several factors can influence the analysis, making the impact of the emissions vary widely. Consideration of these factors is crucial to estimate the ancillary benefits associated with specific mitigation measures.

The modelling of atmospheric concentration reductions of  $PM_{2.5}$  as a consequence of reductions in precursors emissions is a key link in the analysis. Our two approximate methods show results that differ in about 50%. Unfortunately, development of a comprehensive atmospheric model, was outside the scope of this project, and without considerable work may not offer results much better than those of the aggregated models.

Finally, the transference of the unit values from a developed country to a developing one implies some strong assumptions. Until results derived locally became available, this will probably be the weakest part of the analysis.

## **11. Acknowledgments**

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## HEALTH AND ECONOMIC VALUES FOR MORTALITY AND MORBIDITY CASES ASSOCIATED WITH AIR POLLUTION IN BRAZIL

by Ronaldo SERÔA DA MOTTA, Ramon ARIGONI ORTIZ and Sandro DE FREITAS FERREIRA

### 1. Introduction

If divergences on economic values used for valuation exercises are to be accounted among countries, the same concern should be applied for valuation among regions within a country where degree of development varies significantly in regional terms, as it is the case in most developing economies.

Ancillary benefits from mitigation options are key issues to promote actions to combat climate change. Since they are usually locally captured, particularly those related to health benefits associated with air pollution, their valuation require site-specific parameters which may demand a great deal of research and data collection, not always feasible for developing countries.

This study is an attempt to present back-of-the-envelope estimates of morbidity and mortality health benefits associated with air pollution in the Metropolitan Area of São Paulo in Brazil<sup>64</sup>. Atmospheric contamination, mainly caused by mobile sources, is a serious environmental problem in the region and radical changes in transport systems may drastically reduce emissions, including CO<sub>2</sub> ones.

The aim of this study was then to undertake a valuation exercise to offer health cost benefit indicators for air pollution problems in this region applied for evaluation procedures of the new region's transport programme. The study should be an attempt to measure these indicators without relying on direct survey approaches which were not possible within the budget scope of the programme.

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<sup>64</sup>

This paper is based on some results of a health benefit valuation indicators for transport sector in São Paulo (Programa Integrado de Transporte Urbano de São Paulo - PITU) conducted by the São Paulo environmental agency (CETESB) and co-financed by the World Bank. Authors thank all the participants of this research team for comments and suggestions.

Literature on pollution's health costs are prone to suggest several methodological procedures to value health benefits, particularly with emphasis on willingness to pay estimates methods. However, since these methods are costly, several studies in developing countries have applied back-of-the-envelope procedures to account for health costs associated to pollution<sup>65</sup>. More recently, transfer functions have been seen as a promising methodological shortcut to apply WTP based estimates and thus avoiding costly willingness to pay direct surveys<sup>66</sup>.

Therefore, this study applies benefit transfer functions on values estimated for European countries. In addition to that, we also apply short-cut procedures for hedonic and human capital approaches, carried out specifically for the MASP, and discuss the differences of the results. As expected, methodological and data source differences led to great divergence in the results.

The next section briefly presents our estimation procedures and results of each adopted methodology. In our concluding section, we discuss the divergences of the results and their implications for ancillary benefit valuation.

## 2. Methodological procedures

Willingness to pay measures are the basis of environmental monetary valuation. As pointed out in Markandya *et.al* (1999), "the conceptual foundation of all cost estimation is the value of the scarce resources to **individuals**. Thus values are based on individual preferences, and the total value of any resource is the sum of the values of the different individuals involved in the use of the resource. This distinguishes this system of values from one based on 'expert' preferences, or on the preferences of political leaders. The values which are the foundation of the estimation of costs are measured in terms of the **willingness to pay (WTP) by individuals to receive the resource or by the willingness of individuals to accept payment (WTA) to part with the resource**".

Measures of WTP and WTA can be calculated directly through several survey methodological approaches, including contingent valuation, which is the most recommended method. Since these survey oriented approaches are very costly, other methodological options based on indirect valuation are usually employed, such as, marginal productivity losses and surrogated markets<sup>67</sup>.

The most controversial indirect approach is that based on human capital valuation which measures labour output foregone caused by death and morbidity medical care costs. Apart from theoretical problems, its results on output losses cannot be seen as "true" WTP measures.

For the estimation of health cost benefits, hedonic price functions of urban property markets has been largely applied to capture changes in property prices against environmental quality variations across areas. Based on these functions, WTP values are estimated for marginal variations of environmental quality. Data and econometric related problems have, however, made this approach less accepted to calculate full welfare changes.

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<sup>65</sup> See Seroa da Motta, Huber and Ruitenbeek (1999) for a survey of Latin American studies.

<sup>66</sup> See, for example, Markandya (1998) and Pearce (1998).

<sup>67</sup> We are not going to discuss here the several issues related to environmental valuation which are covered by an extensive literature. For those not familiar with the subject, see, for example, textbooks, such as, Pearce and Markandya (1989) and Freeman (1995).

In order to avoid mounting survey costs, it has been recently postulated the application of benefit transfer functions. This estimation procedure relies on the conversion of other sites' WTP measures to a specific area. This conversion is based on differences of social and economic factors which affects the determination of WTP values. As will be seen, for local health benefits such approach may also face serious data availability constraints.

In the following section we present our estimation exercise for the valuation of health effects associated with air pollution concentration in the Metropolitan Area of São Paulo (MASP). This region is by far the most developed area in the country and faces an acute air pollution problem<sup>68</sup>. Our exercise will use short-cut procedures based on the three distinct approaches, as follows:

- Estimating measures of output foregone caused by premature death and health related expenditures.
- Applying benefit transfer functions to European countries' values (ExternE, 1998).
- Adjusting an existing estimate of WTP (Oliveira, 1997) based on hedonic property price function derived for the region.

It must be noted that we are here engaged in an exercise concerning monetary valuations of health risks and not estimates of risk functions for air pollution concentration variations<sup>69</sup>.

### **3. Estimation procedures**

#### **3.1 *Output foregone pricing***

This approach admits that one life lost represents an opportunity cost to society equivalent to present value of its capacity to generate output. Therefore, in the case of a premature death this present value would represent a foregone output which could be taken as a proxy value for the statistical value of life (SVOL).

This approach faces serious criticisms because, as can be seen below, apart from discounting sensitivity, it can be only applied with demographic data, and, consequently, it will use averaging values which precludes people's preferences and risk perceptions. Its results tend, therefore, to offer lower bound WTP estimates.

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<sup>68</sup> See Seroa da Motta and Mendes (1995) for an overview of air pollution problems in Brazil.

<sup>69</sup> For the MASP region, studies on dose-response risk functions associated with air pollution can be found in El Khoury Miraglia (1997).

The present value of future output (PVFO) of a person in the age  $i$  is given by<sup>70</sup>:

$$SVOL \Leftrightarrow PVFO_i = \sum_{j=i+1}^{85} \left[ (P_i^j)_1 \right] \cdot \left[ (P_i^j)_2 \right] \cdot \left[ (P_i^j)_3 \right] \cdot \left[ Y_i \cdot \left( \frac{1+g}{1+r} \right)^{j-i} \right]$$

$\left[ (P_i^j)_1 \right]$  is the probability of a person at the age  $i$  will be alive at the age  $j$ .

$\left[ (P_i^j)_2 \right]$  is the probability of a person at the age  $i$  will be economically active at the age  $j$ .

$\left[ (P_i^j)_3 \right]$  is the probability of a person at the age  $i$  will be working at the age  $j$ .

$g$  is the growth rate of per capita income

$Y_j$  is the expected income of a person at the age  $i$

$r$  is the discount rate.

All the above parameters were taken from demographic surveys conducted by the Brazilian statistical office (IBGE) and income levels from IPEA (1998), relative to the MASP. Since discounting is a crucial parameter in this approach, a sensitivity analysis was taken assuming values of  $r$  of 3 and 10%. Results are presented in Table 1. The resulting average value of PVFO was determined excluding the age brackets over 65 years old which presented the lowest value, as shown in Table 5.

**Table 1. Present value of future output (PVFO) of premature death in MASP (1997 US\$)**

age bracket in years	mortality rate (%)	economically active share	unemployment rate (%)	monthly avg income	PVFO $r = 3\%$	PVFO $r = 10\%$
15-17	0,72	28,70	10,88	217,68	254.777,08	45.195,16
18-24	1,11	65,21	9,23	420,98	255.353,99	60.637,59
25-29	1,11	74,77	5,44	673,39	248.352,34	79.208,47
30-39	2,73	75,33	3,60	870,27	213.299,45	87.066,73
40-49	5,21	72,41	2,13	1.045,46	151.187,34	81.661,27
50-59	7,38	52,21	1,64	971,14	74.100,64	51.871,67
60-64	9,64	29,69	1,25	867,30	24.656,93	19.857,08
65-	12,91	11,25	0,91	860,27	10.959,40	9.325,26

Source: Demographic data from IBGE and income data from IPEA (1998).

<sup>70</sup>

See, for example, Seroa da Motta and Mendes (1995), for a previous application of this approach in Brazil as proposed by Ridker (1967).

For morbidity cases, output foregone estimates are based on observed health expenditures, public and private, which are related to air pollution related disease, namely:

- Medical care costs.
- The respective work days lost.
- Prevention expenditures.

For the purpose of our case, we have considered respiratory disease and heart failure related cases provided by the public health system database (DATASUS) relevant to the MASP.

Although we have faced serious data availability constraints, the following valuation exercise was carried out to estimate these health expenditures. For medical care costs, we have been only able to obtain data on public expenditure related to hospital admissions. Health experts assume that in the MASP, however, private hospitals are covering equal number of cases registered in public attendance for these diseases. Therefore, to account for private hospital cases, we will multiply our public cost estimates by two.

Work days lost were also counted as those related to hospital admissions and measured as the days spent by patients in hospital during the treatment of their respective diseases, as also registered in the DATASUS, multiplied by the average income as reported in IPEA (1998).

**Table 2. Total health expenditure (HE) associated with air pollution in MASP (1997 US\$)**

RESPIRATORY MORBIDITY				
age brackets in years	hospital expenditures	work days lost	monthly avg income	HE
0-14	8.100.408,22	0	0,00	16.200.816,43
15-59	5.989.939,15	141.708	772,10	19.274.036,56
60-	3.736.039,68	79.739	864,23	12.066.302,64
HEART FAILURE MORBIDITY				
age brackets in years	hospital expenditures	work days lost	monthly avg income	HE
0-14	-	0	0,00	-
15-59	10.275.887,10	54.069	772,10	23.334.858,49
60-	12.298.218,09	58.592	864,23	27.972.250,45

Source: Hospital expenditures and work days lost from DATAASUS and income data from (1998).

In order to convert these estimates into equivalent full WTP measures, which should consider the resulting disutility associated with these diseases, we have applied a rule of thumb proposed by Rowe *et al* (1996). This study, based on USA direct estimates, suggests that the ratio of WTP to avoid health risks to medical care costs should be around two. That will result in multiplying again our health expenditures by two. Final results are presented in Table 2 also for age brackets.

Summing up all these expenditures and dividing by the number of hospital admissions for each disease, we calculated our crude estimates on personal health expenditure, proxies to WTP morbidity values<sup>71</sup>, for each disease case, as presented in Table 5.

### 3.2 *Benefit transfer pricing*

Transfer functions are dependent on adjustment variables which affect people's preferences and therefore are based on variables which affect income values among localities.

We have used two transfer functions. A simple one proposed by Markandya (1998), Function 1, based solely on per capita income differential adjusted by purchase parity power index and weighted by the demand income elasticity. We also applied another function, Function 2, proposed by Heintz and Tol (1996) which also includes adjustments for life expectancy and health expenditure variables<sup>72</sup>.

The value of the demand income elasticity ( $e$ ) represents the marginal reduction of a person's WTP value for a certain benefit in relation to a marginal reduction in the person's income and, consequently, it will vary spatially accordingly to changes of people's preferences.

Our exercise will be on transferring European values on health benefits presented in ExternE (1998) to the MASP context.

Again we faced serious data problem. The value of  $e$  for MASP was not possible to measure specifically within the scope of this study and also the other variables are not available to the MASP for the reference year adopted in the European valuation. Therefore, we could only apply national figures for income, expectancy and health expenditures and *ad hoc* values for  $e$ .

Estimation biases are difficult to determine. While one could expect that in MASP purchase power, per capita income, health expenditures and life expectancy would be higher than the assumed values, consequently increasing transferred values, we do not know the bias direction of our assumptions on the parameter  $e$ .

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<sup>71</sup> Here we included the age bracket over 65 years.

<sup>72</sup> Heintz and Tol (1996) do not adjust to purchase power parity as we do in this exercise.

Both functions and data sources are presented below:

Function 1:  $(PPC_{br} / PPC_{eu})^e$

Function 2:  $(PPC_{br}/PPC_{eu})^e \cdot (E_{br}/E_{eu}) \cdot (G_{br}/G_{eu})$

Where:

$PPC_{br}$  = Brazil's per capita income adjusted by purchase power parity (*sources*: IBGE and World Resources, 1998).

$PPC_{eu}$  = European per capita income adjusted by purchase power parity (*source*: Markandya, 1998).

$E$  = national life expectancy (*source*: World Resources, 1998).

$G$  = national health expenditures (*source*: World Resources, 1998).

If variations in personal income and health benefits are valued at par, the value of  $e$  is one. However, Ardila, Quiroga and Vaughan (1998) made specific estimate of  $e$  for Latin American and Caribbean countries based on contingent valuation studies of sanitation programmes which generated a value equal to 0.54. Therefore, due to the sensitivity of this parameter, we have decided to use values of  $e$  equal to 1.00 and 0.54. Results of benefit transfer pricing are shown in Tables 3 and 4.

As can be seen in Table 4, the transfer function factors are very sensitive to the parameters adopted. For example, the introduction of life expectancy and health expenditures results in a value decrease of approximately 25% whereas its combined effect with changes in  $e$ 's value results in reductions of 60%.



**Table 3. Adjustment parameters for benefit transfer functions to Brazil (1995 US\$)**

Parameters	Brazil	Europe
Per capita GDP at Purchase Power Parity (PPC)	5.500,00	17.900,00
Life expectancy (E)	67,1	77,3
Health expenditures (G)	7,4	8,6
Functions	e = 0,54	e = 1
Function 1 (*)	0,528756	0,307263
Function 2 (**)	0,395069	0,229577

Source: Authors' estimates with data from WR (1998) and Markandya (1998).

(\*)  $(PPC_{br}/PPC_{eu})^e$

(\*\*)  $(PPC_{br}/PPC_{eu})^e (E_{br}/E_{eu}) (G_{br}/G_{eu})$

**Table 4. Estimates of transferred values of health benefits to Brazil (1997 US\$)**

			statistical value of life	willingness to pay for hospital admission respiratory morbidity	heart failure morbidity
Europe			4.141.652,32	3.677,52	6.017,76
Brazil	function 1	e = 0,54	2.189.923,05	1.944,51	3.181,93
		e = 1	1.272.574,74	1.129,96	1.849,03
	function 2	e = 0,54	1.636.239,48	1.452,88	2.377,43
		e = 1	950.826,57	844,27	1.381,54

Source: Authors' estimates with European values from ExternE (1998) and Markandya (1998).

### 3.3 Hedonic pricing

Oliveira (1997) using hedonic property price functions estimated WTP values against reductions of particulate matter (PM) concentration in the MASP. Due to data constraints, the author was only able to estimate WTP estimates related to a 10% reductions in particulate matter concentration levels, using only sale prices of new properties. Estimates are calculated in a value range according to distinct econometric functions.

The estimated WTP value represent people's willingness to pay for better air quality in terms of marginal changes of PM. In that case, estimated WTP would cover mortality as well as morbidity risks. Assuming, however, that mortality risk aversion is dominant, we have used this estimate for this purpose as an overestimate value. To make our comparison analysis, we converted the WTP values into equivalent SVOL, using the expression  $SVOL = WTP / \Delta R$  where  $\Delta R$  is the risk factor<sup>73</sup>, indicating maximum and minimum figures. The value of  $\Delta R$  was taken as 0.006 from Ponka *et al* (1998) for both diseases.

#### 4. Conclusions

As said before, if divergence on WTP values are to be accounted among countries, the same concern should be applied for valuation among regions within a country with significant variations on factors and parameters affecting WTP, as it is the case in most developing economies.

Our exercise applying short-cut valuation approaches has, however, shown that data constraints are dominant in each of them when one is willing to estimate site-specific values for ancillary benefits, as in our case for the Metropolitan Area of São Paulo (MASP),

In Table 5 we summarize our estimates and, as can be seen, they vary significantly according to each methodological approach. As expected, mortality results are the highest from benefit transfer valuation whereas output foregone estimates are the lowest ones. Although the hedonic pricing estimate of SVOL is just in the middle of these two other estimates, one must bear in mind that it also includes morbidity risks.

As already mentioned, due to the introduction of adjustment parameters, divergences within transfer benefit estimates are almost in the order of 3. When comparison is made with other approaches, for example, in the case of SVOL, variation may reach the factor of 30 between output foregone and benefit transfer pricing. Although for the case of morbidity WTP, results for these two approaches tend to be closer for respiratory diseases, they are again quite divergent in values for heart failure.

The results above have emphasized that, apart from the methodological divergences, there are also serious data source constraints if one is trying to make site-specific calculations.

Difficulties associated with the need for comprehensive data on property and air pollution concentrations were faced in the reported survey on hedonic pricing. For benefit transfer functions, the adjustment parameters were not available for the MASP.

In the case of output foregone pricing, although all required demographic data was available for output losses, its estimates are very controversial and highly dependent on discounting.

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<sup>73</sup> See, for example, Markandya (1998) and Pearce (1998).

**Table 5. Summary of the estimates of health benefit values associated with air pollution in MASP (1997 US\$)**

<b>Valuation of Statistical Life</b>		
<b>Transfer Pricing</b>		
function 1 - e=0,54		2.189.923,05
function 1 - e=1,00		1.272.574,74
function 2 - e=0,54		1.636.239,48
function 2 - e=1,00		950.826,57
<b>Hedonic Pricing</b>		
minimum		166.000,00
maximum		487.406,67
<b>Output Foregone Pricing</b>		
r=3%		197.664,07
r=10%		73.079,05
<b>Willingness to Pay for Morbidity Risk Reduction</b>		
	RESPIRATORY	HEART FAILURE
<b>Transfer Pricing</b>		
function 1 - e=0,54	1.944,51	3.181,93
function 1 - e=1,00	1.129,96	1.849,03
function 2 - e=0,54	1.452,88	2.377,43
function 2 - e=1,00	844,27	1.381,54
<b>Hospital Expenditures</b>	1.985,79	7.336,85

However, as already emphasized, our aim with this exercise was not to verify convergence in results derived from distinct valuation approaches which have been fully explored in the relevant literature<sup>74</sup>. The main message here is to discuss the possibility of applying these short-cut approaches to offer reliable economic indicators for ancillary health benefits.

Our intention was to show that, if health benefit measures are important ancillary benefits to justify and promote actions to combat climate change, more research efforts should be devoted to their measurement, since the choice of any one of these specific short-cut approaches will significantly affect the economic assessment of these actions. Consequently, it seems that WTP surveys must be promoted and improved in developing countries to offer reliable health benefit valuations. This is another opportunity for north-south research cooperation in the field of climate change issues.

<sup>74</sup> For example, ExternE (1998) offers a comprehensive on this matter.

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# EVALUATION OF HEALTH EFFECTS OF POLLUTION

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## 1. Introduction

Current initiatives to improve air quality in the Mexico City Metropolitan Area (MCMA) require estimation of the economic evaluation of the benefits gained from proposed programs. This document presents a review of the knowledge of health effects and more specifically a meta-analysis to summarise data available and obtain an estimate of exposure-response relations to be used to predict the number of health events that could be avoided by improving air quality.

This overview is restricted to particulate matter and ozone because these are the pollutants of more concern in this megalopolis. The first section presents an overview of the toxicology and exposure to air pollution, followed by a meta-analysis of published international and Mexican studies. The review was based on recent epidemiological studies of the association of acute and chronic exposures to particulate air pollution or ozone with increased morbidity and mortality. Specific health effects include acute effects on mortality, hospital emergency room visits, respiratory symptoms, restricted activity days, as well as the chronic effect on mortality and respiratory symptoms. To obtain an estimated average, studies were pooled using random effects models. These models take into account between study variability as a result from among sampling sites and the variance within the studies. Exposure-response curves are presented as increases in relative risks per  $10\mu\text{g}/\text{m}^3$  in PM10 and 10 ppb in ozone.

## 2. General overview of air pollution and health

Anthropogenic air pollution has been a way of life for almost 500 years now. The industrial revolution introduced great strides in technology, society and services; however, it also initiated the production of huge quantities of pollutants emitted into the air with no notion of how they might affect health. At the time, smoke from burning coal was the major pollutant, but this was only the beginning of countless air pollutants which have since proven harmful to human health (Dockery and Arden Pope 1996). Since that time, many episodes have been recorded where elevated levels of pollutants have caused serious health effects in different populations. One of the most well-known cases occurred in London in December, 1952, when environmental conditions caused a 5-day accumulation of air pollution, especially sulphur dioxide and smoke, reaching  $1500\text{ mg}/\text{m}^3$  and resulting in an increase in the number of deaths to around 4000 for the period. In New York City in 1963, conditions similar to those occurring in London caused 400 deaths. These cities are not alone reporting such events. High levels of air pollution have been registered in Mexico City, Rio de Janeiro, Milan, Ankara, Melbourne, Tokyo and Moscow, to name only a few problematic cities (Dockery and Arden Pope 1996).

Since major cities frequently suffer episodes of severe pollution, they require special surveillance to protect the large number of individuals concentrated there and the important economic activities carried out therein. It is precisely due to the flourishing economic activity in these areas that the environment has been relegated to secondary importance. On the other hand, different diseases, from respiratory to cardiac ailments, in different degrees of severity from minor irritation to death, have been associated to exposure to air pollution (Dockery and Arden Pope 1996). Some of the more important toxic effects will be described in the following chapters of this report.

## **2.1 Sources of exposure**

The majority of substances considered as environmental pollutants are produced through human activities such as the use of internal combustion engines (automobiles), power plants and industrial machinery. Because these activities are performed on such large scale, they are by far the major contributors of air pollution, with cars estimated as responsible for approximately 80% of today's pollution. Minor sources of pollution such as lawn mowers, cooking stoves, stationary diesel fuel tanks, heaters, gasoline stations, laundries, other cleaning services, etc. are currently being evaluated as well (Möller *et al.* 1994, Pooley *et al.* 1999).

All the exposure sources mentioned above can be classified as anthropogenic. Natural sources of pollution include soil erosion (the wind carries airborne particulate matter produced through erosion), evaporation of sea water (which carries with it various materials), volcanic eruptions and forest fires (which send toxic substances directly into the atmosphere) (Pooley *et al.* 1999).

## **2.2 Classification of environmental pollutants**

We now know that air pollution is a complex mixture of a variety of substances produced by incomplete combustion reactions mainly resulting from anthropogenic activities but also through natural phenomena. Pollutants can be classified in a variety of ways. Table 1 shows some classifications based mainly on physical and/or chemical properties.

Table 1. **Classification of environmental pollutants**

<b>1) Chemistry</b>	
a) Inorganic	For example: sulphates ( $\text{SO}_4^{-2}$ ), nitrates ( $\text{NO}_3^-$ ), ammonium ( $\text{NH}_4^+$ ), sulphur oxides ( $\text{SO}_x$ ) and elemental carbon, which can form salts with: Fe, Mn, Zn, Pb, V, Cr, Ni, Cu, Co, Hg and Cd, and even with As and Se.
b) Organic	For example: benzene, 1-3 butadiene, polycyclic aromatic hydrocarbons, dioxins, CO and $\text{CO}_2$ .
<b>2) Source</b>	
a) Primary:	Pollutants are emitted directly into the atmosphere
b) Secondary:	Pollutants are emitted as supersaturated gasses and in the atmosphere become solid or react to form a different species (this phenomenon occurs mainly with polar compounds).
<b>3) Physical Nature</b>	
a) Dust	Particles produced by mechanical disintegration of solids.
b) Aerosol	Suspension of solids in the air, particles can be 1 nm to 2 $\mu\text{m}$ in diameter, capable of remaining suspended in the air and moving easily.
c) Smoke	Material produced by the incomplete combustion of organic substances, generally of small particle size ( $\leq 15\mu\text{m}$ ).
d) Black Smoke	Non reflective particulate matter.
e) Vapor	Condensation product of evaporated material (iron oxides) and smoke.
<b>4) Particle size</b>	
Ultra fine particles (0.01-0.1 $\mu\text{m}$ )	These are produced from supersaturated gasses such as $\text{SO}_2$ , $\text{NH}_3$ and $\text{NO}_x$ ,
Fine particles (accumulation) (0.1-0.25 $\mu\text{m}$ )	These are composed generally of $\text{SO}_4^{-2}$ , $\text{NH}_4^+$ and $\text{NO}_3^-$ , do not settle to the ground and are capable of travelling long distances.
Rapidly settling Particles (1-20 $\mu\text{m}$ )	
Large particles (>20 $\mu\text{m}$ ),	Among these are the soil particles and some metallic salts with Al, Fe, Mn, Sr, Ca, Co and K

Source: Information from Möller et al. 1994, Wilson et al. 1996, Ghio et al. 1999, Pooley et al. 1999.

## 2.3 Toxicology of air pollutants

### 2.3.1 Relationship between the toxic effect and physical and chemical properties of air pollutants

Not all air pollutants have the same capacity for producing toxic effects, nor do they cause the same damage. It is a logical conclusion that the differences are due to the physical and chemical properties of these components. This report will briefly mention the properties as they relate to toxicity.



Beginning with the molecular aggregation state, substances in aerosol form have been shown to be more toxic than compounds in gaseous state. This is due to the fact that gaseous compounds are eliminated by the respiratory system much more easily than aerosols, which are rapidly deposited or absorbed. The particle size of an aerosol, between 1 nm and 2  $\mu\text{m}$ , is easily deposited in the respiratory system (Wilson *et al.* 1996).

Particle size determines the extent to which the particles can penetrate into the respiratory system. Table 2 shows penetration ability of particles as a function of size. Once particles have entered the respiratory tract, depending on their size they can accumulate in different sites within the respiratory system. The major regions of accumulation are extrathoracic (nostrils and larynx), bronchial (trachea, bronchial and terminal bronchial) and alveolar (bronchiole and alveolar sacs). Up to 50% of particles smaller than 0.02  $\mu\text{m}$  can be deposited in the lungs (ICRP 1996, Ghio *et al.* 1999).

Table 2. Particle penetrability according to size

Particle size	Region to which penetration can occur
> 11 $\mu\text{m}$	Captured in the nostrils, do not penetrate into the lower respiratory tract.
7-11 $\mu\text{m}$	Nasal passage
4.7-7 $\mu\text{m}$	Larynx region
3.3-4.7 $\mu\text{m}$	Trachea and primary bronchial region
2.1-3.3 $\mu\text{m}$	Secondary bronchial section
1.1-2.1 $\mu\text{m}$	Terminal bronchial section
0.65-1.1 $\mu\text{m}$	Bronchioles
0.43-0.65 $\mu\text{m}$	Alveolar

Source: Information from Wilson *et al.* 1996 y Ghio *et al.* 1999.

Toxicity of environmental pollutants has also been related to chemical reactivity as acids or alkalis. The most studied compounds from each group are  $\text{NH}_3$  and  $\text{NH}_4^+$  (alkalis) and  $\text{SO}_2$  and  $\text{H}_2\text{SO}_4$  (acids). Of these,  $\text{H}_2\text{SO}_4$  has been shown to be the most toxic. Both acids and bases can be found in the same region of the atmosphere where they combine to produce neutral species. However, when alkalis are present in greater abundance than acids, they tend to produce more severe toxic effects than when acid species dominate (Schlesinger *et al.* 1995, SUH 1995). In a 1995 German study of respiratory disorders, most were not attributable to the presence of acidic molecular pollutants (Brauer *et al.* 1995).

Another property of chemical elements, mainly the metals, is their ability to convert to other species by oxidation or reduction of other components present in the atmosphere or within the organism, itself (redox potential). This type of reactivity is associated with certain effects such as neutrophilic alveolitis, hypersensitivity reactions, increased lung infections and death. The substance must be in solution for these oxidation-reduction reactions to occur, and for this reason, the more soluble salts have greater toxic potential. This solubility-toxic relationship persists as well for non-metallic compounds such as  $\text{NH}_4\text{HSO}_4$ ,  $(\text{NH}_4)_2\text{SO}_4$  and  $\text{H}_2\text{SO}_4$  (Wilson *et al.* 1996, Ghio *et al.* 1999).

## 2.4 Toxic effects of air pollutants

Chemical compounds emitted into the atmosphere due to human activity or those compounds that are byproducts of the interaction of chemical emissions have been shown to have adverse effects on health. These effects, as discussed in this report, depend fundamentally on the nature of the compound in question, the concentration in the air and the time of individual exposure. Noxious health effects caused by air pollution can be classified as due to either chronic or acute exposure.

### 2.4.1 Health effects due to acute exposure to air pollutants

Toxic effects attributable to acute exposure to air pollutants vary widely and have been reported practically since the beginning of the industrial revolution where episodes of high levels of pollutants were associated with increases in diverse respiratory and heart diseases and death. These episodes have occurred on more than a single occasion in different parts of the world, especially in highly industrialised and/or populated areas (Ellison & Waller 1978, Holand *et al.* 1979, SMY 1979, Bates 1980).

The most studied toxic effect due to acute exposure to environmental pollutants is mortality. Many reports describe an increase in total mortality (not including accidental death) associated mainly with exposure to particulate matter (PM), ozone and sulphates. This association can be disputed, however, since the cause of death should be related to the route of exposure (Schwartz 1994<sup>a</sup>, Dockery and Pope 1994).

A great number of studies report increases in mortality due to respiratory complications, and in this case, the mechanism obviously can be related to exposure to air pollution. Many reports also claim an increase in death due to cardiovascular ailments, which would implicate a mechanism with an indirect effect from air pollution. Both causes of death are associated with exposure mainly to PM, ozone and sulphates. Mortality attributable to exposure to air pollution occurs mainly in individuals who suffer from cardiac and/or respiratory diseases. Increased mortality in these groups occurs within 1 to 5 days following the hazardous exposure (Schwartz 1994<sup>a</sup>, Wilson *et al.* 1996, Cropper, L. (1999).

Certain population groups are more susceptible than others to the effects of pollution, which has attracted the special attention of many researchers in the field. Individuals at the extremes of the life cycle, the elderly and infants, show increased mortality associated with exposure mainly to PM and sulphates. Although the mechanisms leading to death are the same as those causing toxic effects, these groups' biological defence mechanisms are less efficient than in the rest of the population.

Increased mortality due to exposure to air pollutants can also be associated to smoking habits. This phenomenon is likely due to the fact that smokers have a 30% decreased lung capacity compared to non-smokers of the same age (Wilson *et al.* 1996).

Besides mortality, a great number of acute conditions have been reported associated to exposure to air pollutants. Among these are diseases of the respiratory tract, both upper and lower, bronchitis, pneumonia, chronic obstructive pulmonary disease and cough with phlegm.

The proposed mechanism producing such diseases could be related to the ability of certain pollutants to produce systemic ( $\text{NO}_2$ ) and local immunosuppression. In exposed animals ( $\text{SO}_4^{2-}$  and  $\text{NO}_3^-$ ), a decrease in the affinity of macrophages for the Fc section of antibodies has been observed. Intuitively, in a human organism with diminished immune response, the capability to mount an adequate defence in a populated, urban environment, where exposure to multiple pathogens is high, would be unfavourable (Schlesinger *et al.* 1995, Ehrlich 1980).

Along this same line, many laboratory animal studies have evaluated the effects of pollutants on macrophages, one of the major cellular defence lineages present in the respiratory apparatus. Two types of effects have been observed. Exposure to certain pollutants ( $\text{SO}_4^{2-}$  ó  $\text{NO}_3^-$ , for example) causes a decrease in affinity for the Fc section of the immunoglobulins and limits the antibody mediated response. In addition, exposure to transition metals results in increased secretion of reactive intermediates of oxygen ( $\text{O}_2^-$ ,  $\text{OH}^-$  and  $\text{H}_2\text{O}_2$ ) and nitrogen ( $\text{NO}^-$  and  $\text{ONOO}^-$ ), producing a state of tissue inflammation. It is possible that other cytokines, such as some of the interleukins, are affected, as well (Schlesinger *et al.* 1995, Wilson *et al.* 1996, Martin *et al.* 1997, Ghio *et al.* 1999).

However, environmental pollutants likely also effect somatic cells directly. Exposure to  $(\text{NH}_4)_2\text{SO}_4$  and  $\text{NH}_4\text{NO}_3$  has been shown to increase lung tissue permeability, leading to saturation of the intercellular spaces with interstitial fluid. This could lead to pulmonary edema or a chronic inflammatory state, decreasing gas exchange in the lungs and resulting in a hypoxic state (Kleinman *et al.* 1995, Schlesinger *et al.* 1995).

Thus far, we have mentioned only diseases which develop favoured by exposure to air pollution. Other symptoms are exacerbated as well by exposure to certain pollutants such as ozone and PM, which are associated with increased asthmatic attacks, coughs without phlegm and wheezing (Pope *et al.* 1991, Roemer *et al.* 1993).

The mechanism by which these symptoms are increased could be related to effects on the immune system. Although the cause remains obscure, ozone, sulphates and PM can stimulate over induction of immunoglobulins, such as IgE, which initiates a series of signals resulting in the production of spasms of certain muscle groups (Wilson *et al.* 1996, Ghio *et al.* 1999).

#### 2.4.2 Health effects due to chronic exposure to air pollutants

Pollution episodes, which have occurred in different cities around the world, have demonstrated the consequence of human exposure to high concentrations of air pollution. However, these episodes appear sporadically, and currently, exposure to low concentrations of pollutants over long periods of time is a daily phenomenon. Recent studies have focussed on establishing the effects of chronic exposure over prolonged periods.

A synthesis of all the available information concerning chronic exposure is an extremely complex task due to the enormous number of factors which could be associated with the same types of symptoms, such as active and passive smoking, nutritional level, etc. It is very difficult to establish a single causal agent which could be responsible for a cancer, for instance, since this type of disease develops over a long period of time and involves various interacting factors (Möller *et al.* 1994, Schlesinger *et al.* 1995).

Health effects due to chronic exposure are very similar to those reported for acute exposure. There are several reports of increased mortality, however, most cases involve mainly elderly individuals where respiratory and cardiovascular problems are already the principle cause of death (Anderson 1996, Borja 1997, Pope 1996).

Increased respiratory diseases (such as bronchitis) have also been reported associated to chronic exposure. The mechanisms causing these diseases should be very similar to those occurring for acute exposure.

The best documented chronic effect of exposure to air pollution is cancer. Approximately 70 to 80% of all cancer types have been reported as due to exposure to environmental pollutants. The mutagenic properties of different substances (e.g. diesel) have been demonstrated, and, as we well know, mutation is an essential step in the transformation of a normal cell to a cancerous cell. The mutagenic ability of a substance is not the only property that can stimulate cell transformation, however. Over-activation or inhibition of regulatory enzymes can also lead to cellular transformation.

A chronic inflammatory state can also lead to cancer development. Exposure to some environmental pollutants (transition metals) can result in a chronic inflammatory state due to altered secretion of reactive intermediaries of oxygen ( $O_2^-$ , OH and  $H_2O_2$ ) and nitrogen (NO and ONOO), possibly induced by increased secretion of a cytokine that induces the production of these reactive intermediaries and the activation of macrophages long-term result of a continuous inflammatory state can result in tissue lesions and even cancer (Martin *et al.* 1997).

For both cases of chronic and acute exposure to air pollutants, populations are exposed to a complex mixture of compounds whose combined toxic effects could differ from that of each isolated compound. In a study performed on volunteers who were exposed to ozone with and without pre-exposure to  $H_2SO_4$ , the pre-exposed group suffered more severe toxic effects than the group that was not pre-exposed (Thurston and Ito 1999).

Other mixtures that have proven more toxic than the individual compounds include  $SO_2$  - ozone,  $SO_2$  - black smoke and  $PM_{10}$  - ozone (Katsouyanni 1995). It is therefore necessary to develop models and protocols to analyse the different interactions among environmental pollutants (Samet *et al.* 1993).

## 2.5 *PM<sub>10</sub> particles*

In the field of environmental pollution toxicology, much interest has been recently shown in the study of  $PM_{10}$  and  $PM_{2.5}$  particles. These particles are associated with diverse respiratory system pathologies and they contribute to indoor exposure, since their size allows them to penetrate interior spaces.  $PM_{10}$  and  $PM_{2.5}$  particles are defined as a mixture of different compounds with 50% of the solid material able to pass through a 10  $\mu m$  ( $PM_{10}$ ) or 2.5  $\mu m$  ( $PM_{2.5}$ ) sieve (Koutrakis and Sioutas 1996).

Among the different  $PM_{10}$  and  $PM_{2.5}$  components are organic compounds, such as benzene, 1-3 butadiene, polycyclic aromatic hydrocarbons, dioxins, etc., inorganic compounds, such as carbon, sulphates, nitrates, chlorides and even some metals (Wilson *et al.* 1996, Pooley *et al.* 1999).

The particles produce toxic effects according to their chemical and physical properties, as described above. However, they primarily affect susceptible individuals, where their effects are much more severe than those produced in normal individuals (Schlesinger *et al.* 1995, Toster 1999).

Due to the size of the PM<sub>10</sub> and PM<sub>2.5</sub> particles, their half-life in the atmosphere is generally very high since they do not settle to the ground but remain suspended and can be transported very far from their origin. This property is very important to consider since a population far from the pollution emission site may be exposed to the same extent as one close by (Wilson *et al.* 1996).

## 2.6 Ozone

Ozone is poorly soluble but highly reactive gas, is mainly produced in the troposphere by series of sunlight-driven reactions involving nitric oxides and volatile organic compounds. It is partially depleted in the upper airways when inhaled but a major fraction does reach the lower airways. Ozone can react with uric acid, which is secreted by human airway's submucosal glands and is present in near mmol/l concentrations in nasal surface liquid. Pryor and his colleagues have proposed that some of the toxic products of the latter reaction (hydroxyhydroperoxides, hydroxyaldehydes) are important mediators of ozone effects on underlying epithelium and some scientists have calculated that ozone per se does not even reach the epithelial cell apical membrane in conducting airways (Bromberg 1999).

The proportion of ozone uptake attributed to surface liquid decreases progressively as the surface liquid thins and/or its reactivity with ozone diminishes. Accordingly, the highest epithelial tissue dose is predicted for the terminal bronchiole-respiratory bronchiole region. This is indeed a site of damage in ozone-exposed animals. Bronchoscopic sampling along airways also indicates that a substantial fraction (35%) of orally inspired ozone is taken up in the upper airway and trachea and that ozone in exhaled air is limited to the initially expired volume representing airways dead space (Bromberg 1999).

That inhalation produces toxicity in large airways is supported by evidence of ciliated cell loss and increased epithelial mitotic index in small animals, neutrophilic inflammation in humans, increased bronchial artery blood flow in sheep and by the symptoms of cough and of substernal pain exacerbated by deep inspiration in humans (Bromberg 1999).

## 2.7 Populations at risk

Every individual has a different susceptibility to air pollutants. The level of individual risk is defined by genetics and biology, age (especially vulnerable are those individuals at the life cycle extremes), nutritional state, presence and severity of respiratory and cardiac conditions and the use of medications (Wilson *et al.* 1996). A good example of varying individual risks is demonstrated by a study evaluating maximum expiratory flow in healthy children, children with minor respiratory disease and those with asthma, with and without pharmacological treatment, all exposed to various environmental pollutants. The results showed an association between exposure and disease only in children with asthma who were under pharmacological treatment, in other words, those children who were most seriously ill (Roemer *et al.* 1999). Similar studies showed that adolescents suffering from asthma are extremely sensitive to exposure to SO<sub>2</sub> (Speker 1999).

Other susceptibility factors that could be associated with respiratory diseases are the presence of certain alleles (genetic susceptibility), enzymatic isotypes involved in the metabolism of environmental xenotoxins (such as members of the cytochrome P-450 family, glutathione s-transferase), and enzymes involved in the DNA repair process (Möller *et al.* 1994). Age is also an important factor, with preadolescents (< 13 years) and the elderly (> 65) at greatest risk (Wilson *et al.* 1996, Ghio *et al.* 1999).

## 2.8 *Air pollution exposure factors*

The major sources of human exposure to air pollution are, as mentioned above, those produced by human activity. Pollutants can enter the organism in various ways such as ingestion, absorption through the skin and inhalation (Möller *et al.* 1994, Wilson *et al.* 1996). Inhalation is the major route of entry for exposure to air pollution. An important aspect of inhalation that is often ignored is oral breathing. When individuals breathe through the mouth, the physical and mechanical barriers of nasal breathing are absent, and oral breathing has been shown to decrease the ability to eliminate particles deposited in the respiratory tract, mainly in the upper air ways (Wilson *et al.* 1996).

Until recently, only outdoor areas (exterior) were considered as exposure sites since that was where an individual would contact the majority of air pollutants. We now know that this is true only for certain types of pollutants such as metals, which due to their particle size are found essentially only outdoors (this is true for any particulate pollutant with a particle diameter greater than 10 µm). Carbon monoxide (CO) and nitrogen dioxide (NO<sub>2</sub>), on the other hand are found in greater quantity indoors (Möller *et al.* 1994, Maynard 1999).

A study in the United States showed that individuals spend an average of 87.2% of their time indoors, 5.6% of their time outdoors and 7.2% in transit (Wilson *et al.* 1996), and values for Mexico are 83.7%, 11.50% and 0.05% correspondingly (Rojas-Bracho 1994). These data demonstrate the importance of determining indoor, as well as outdoor, exposure when precisely defining an individual's true exposure.

Other factors must also be considered when determining exposure. The degree of dispersion or accumulation of contaminants depends on weather conditions. An increase in temperature, for instance, provides convection currents that help to disperse pollutants (Brauer *et al.* 1995), although some studies claim more respiratory problems reported on warmer days (Katsouyanni 1995). Mexico City is recognised world wide as a prime example of where geographical and weather characteristics play an important role in pollution accumulation. The conditions in Mexico City generally favour accumulation of pollutants (Programa para mejorar la calidad del aire en el valle de México 1995-2000).

All these factors must be taken into consideration when establishing exposure levels to environmental pollutants. This requires a fractionated evaluation where pollutants in the microenvironment, the time the individual spends in this environment as well as other factors which could confuse a precise evaluation of exposure are all considered (Möller *et al.* 1994, Wilson *et al.* 1996). Over all, establishing exposure to environmental pollutants for an urban dwelling individual is extremely complex (Möller *et al.* 1994, Wilson *et al.* 1996).

## 3. **Meta-analysis of human health effects of particulate matter and ozone**

In order to evaluate the health risks and costs due to air pollution (specifically ozone and PM<sub>10</sub>) in the Mexico City Metropolitan Area (MCMA), we required estimates of the changes in incidence of adverse health effects associated with projected changes in air quality. Estimates of the changes in air quality and the population exposed are presented in another section of this report. This section presents the method used to derive the concentration-response functions.

The number of published studies of the health effects of air pollution has grown during the last decade; however, specific studies in the MCMA are still limited. Therefore, we decided to summarise international and national published relevant reports via a meta-analysis. The methodology of this analysis focuses on combining the results from the various studies to identify consistent patterns. Due to the rapid growth of the field of epidemiology since the 1960's, the number of publications is overwhelming and the classical narrative review is no longer appropriate for summarising findings in this field. . Meta-analysis of published papers has several limitations. Heterogeneity (including confounding) and publication bias are among the most important. Pooled estimates should be taken with caution if heterogeneity between studies is high, sensitivity analysis would be preferable (Blettner,1999). Conventional statistical analysis with fixed effects, that is to assume only sampling error in studies, do not take into heterogeneity resulted from sampling sites differences. When heterogeneity is present, random models incorporate variation between the studies, assuming that each study has its own true exposure effects and that there is a random distribution of the true exposure effects around a central effect. However, if we presume heterogeneity, the use of random effects is limited too, since it is not sufficient to explain the heterogeneity between studies, since the random effect merely quantifies unexplained statistical variation. Heterogeneity between studies should yield careful investigation of the sources of the differences, i.e. population characteristics, household conditions, particles composition, statistical models used, control of confounders etc. Since information on relevant characteristics like particulate composition was not available for Mexico City, and due to time constraints we decided to reduce heterogeneity with the inclusion criteria and use the between-study variance to weight the studies with random effects models.

### **3.1 Methods**

#### *3.1.1 Identification of publications*

The first step in this analysis was an exhaustive search of published studies on human health effects due to exposure to ozone and PM<sub>10</sub> via Medline, Pubmed, Biomed-net and Aries databases. Manual library searches were also performed examining particularly Mexican publications. Besides providing a general theoretical structure for the analysis, these search results served to compile a summary of the major toxicology aspects of environmental pollution.

No results of laboratory animal studies were included in the analysis because of the difficulty to extrapolate results to environmentally exposed humans. Human populations exposure occurs with a variety of diseases and different severity levels, unlike most laboratory animal studies, which are performed using healthy animals. Humans are usually exposed to several pollutants simultaneously while most animal studies deal with exposure to a single compound. In addition, humans are normally exposed to chronic doses of pollutants while animals are subjected to acute or sub-acute exposure, and obviously the biological responses to the same chemical varies for different species (Kodanvanti 1999).

## Selection and Classification of Material

Not all the bibliographic material collected was used in the statistical analysis. Criteria of inclusion was: a) peer-reviewed published papers evaluating the association between exposure to ozone or particles and clinically identifiable human health effect (biochemical and molecular effects were not included), b) quantification of any type of particles, Total suspended particles (TSP), black smoke (BS), Coefficient of haze (CoH) or any PM. Criteria for exclusion was: a) papers not presenting information for the variance, standard error or confidence intervals for the association estimate (percent change, RR or OR), b) reports based on small populations, c) absence of control for temperature and seasonal variation over the study time period. In order to separate the effects of particles and ozone, specially mortality, we classified the studies that used multivariate models to take into account spatial and time correlation of these pollutants.

### 3.1.2 Air pollutants

Reports were classified according to location and time period as well as average and range of PM<sub>10</sub> and ozone levels. For studies covering a period of several years, annual averages were used, and for shorter studies of one year or non-continuous time periods, pollutant averages given by the authors recorded. For ozone studies, if possible the average maximum for one hour was used. If this value was not available, the author's reported value recorded.

Not all articles provided PM<sub>10</sub> data since for each case this depended on the method used for particle quantification. Usually the particles were reported as total suspended particles (TSP) including black smoke (BS), PM<sub>15</sub>, PM<sub>13</sub>, PM<sub>10</sub>, PM<sub>7</sub>, PM<sub>2.5</sub> or the Haze coefficient (CoH). In order to produce homogeneous results in terms of PM<sub>10</sub>, the following table of approximate equivalencies was used.

Table 3. **Approximate Equivalencies PM<sub>10</sub>**

---

$PM_{10} \cong PM_{15}$
$PM_{10} \cong PM_{13}$
$PM_{10} \cong TSP * 0.55$
$PM_{10} \cong PM_{2.5} / 0.6$
$PM_{10} \cong CoH / 0.55$
$PM_{10} \cong BS$

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Source: Dockery et al. 1994.



### 3.1.3 Information on health effects

The analysis included all health effects reported for human populations. These included total mortality, mortality due to respiratory causes, mortality due to cardiovascular causes, mortality in individuals above 65 years of age, child mortality, total hospitalisations, hospitalisations due to respiratory causes, hospitalisations due to cardiovascular causes, emergency room attendance, emergency room attendance for respiratory causes, emergency room attendance for cardiovascular causes, all effects reported for asthmatic individuals, all effects reported for asthmatic individuals using bronchial dilators, effects on functional respiratory parameters (FVC, FEV-1, etc) and all respiratory effects reported for the general population. However, for the purpose of the final analysis only non-overlapping health effects are to be used in order to avoid double counting of benefits from overlapping endpoints. For example, the literature reports relationships for hospital admissions for single respiratory ailments, as well as for all respiratory ailments combined.

### 3.1.4 Concentration response functions

Most studies express the health effect ( $y$ ) a function of an amount of change air pollutant level ( $\Delta AP$ ). The calculation of the corresponding ( $\Delta y$ ) depends on a C-R function from epidemiological studies. The C-R estimated in these studies may differ from each other in several ways, standard definitions of health endpoints, baseline populations and the shape of the relationship. Some studies assume linear relationships, while others log-linear functions. The linear relationship is of the form  $y = \alpha + \beta P$ . The log linear relationship is of the form:  $y = \beta e^{\beta P}$  or, equivalently  $\ln(y) = \alpha + \beta P$ . Despite some statistical limitations, results from different studies were transformed to represent percent changes in the health effect for each 10 units of variation in the pollutant concentration. Since authors reported values in different C-R functional forms as odds risk (OR), relative risk (RR), percent increase, and regression results or coefficients of regressions, we used the following transformations:

- For RR or OR: RR or OR value was subtracted 1 and from the result multiplied by 100. This operation converted the units to percentages of the health effect. Each quantity was then divided by the value of change according to the concentration used by the author in the article. These values could be some percentile rank, maximum value, average, 100 units of concentration, 50 units of concentration, etc. When the author used continuous variables, the RR or surrogate was multiplied by 10 to provide a percent change for 10 units of concentration.
- For percent change. In this case, the percent change was divided by the value of concentration used by the author in the article. When the author used continuous variables, the RR or surrogate was multiplied by 10 to provide a percent change for 10 units of concentration.
- For coefficients (Poisson or logarithmic). First we determined whether this coefficient had been multiplied by some unifying factor (usually done to simplify notation). If so, the original value was recovered through the appropriate operations, as indicated for each table in the methods or results. The original value was then multiplied by 1000 to convert  $\beta$  into a percentage for 10 units of concentration ( $100 \times 10 = 1000$ ).

To calculate the confidence interval, one of the following two procedures was used:

- If confidence intervals were reported in the article (these are commonly included for RR and OR), the same adjustment was made as for RR or OR, accordingly.
- If the results were given in terms of a regression coefficient or when no confidence interval was reported, the author usually provided a value for standard error. In this case the adjustment was made as if for a regression coefficient and then added and subtracted to the main value to provide intervals.

To simplify all this information graphical presentation was prepared for each health effect.

### *3.1.5 Pooled estimate*

To obtain a single pooled estimate of the health effects reported from the selected studies a weighted average was used. C-R functions were weighted according to the statistical precision of the studies and the between-studies variance, using random effects models. Since the proposed mother project will be carried out in Mexico City, articles based on Mexico City population were given double the weight of international cases, because they are thought to reflect more the Mexican reality in terms of susceptibility and sociodemographic characteristics. For pooled estimate are presented with confidence intervals at 95%.

## **4. Results**

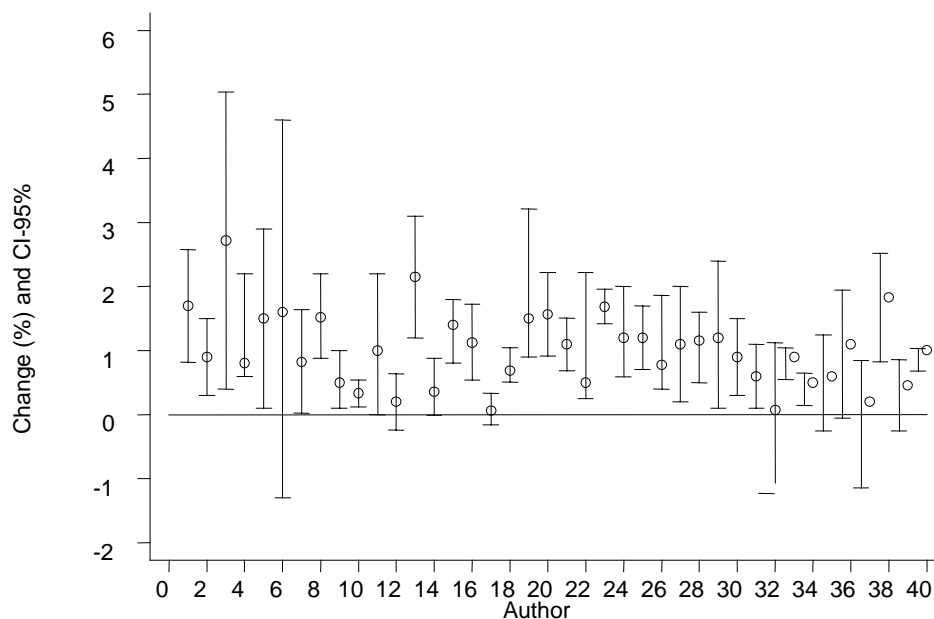
We performed an extensive meta-analysis with the most current national and international literature describing the effects of air pollution (specifically ozone and  $PM_{10}$ ) on human health, with aim to characterise in a ecosystemic point of view, the contamination health risk, the magnitude of the damage and the cost on the human health. The report below summarises this review with the latest available information on this topic.

### *4.1 Meta-analysis of health effects caused by exposure to $PM_{10}$*

#### *a) Percent change in mortality due to exposure to $PM_{10}$*

Of all the toxic effects attributed to  $PM_{10}$ , death has been the most thoroughly documented. Death due to effects of air pollution occurs generally between 1 and 5 days after the hazardous exposure. Since the 1950's, many studies have recorded increased mortality associated with high levels of pollution. In this analysis, we have included the major studies carried out in the Americas, Europe, Australian and Asia since 1970.

Figure 1. Percent change in general, non-accidental mortality for each 10 µg/m<sup>3</sup> increase in PM<sub>10</sub>



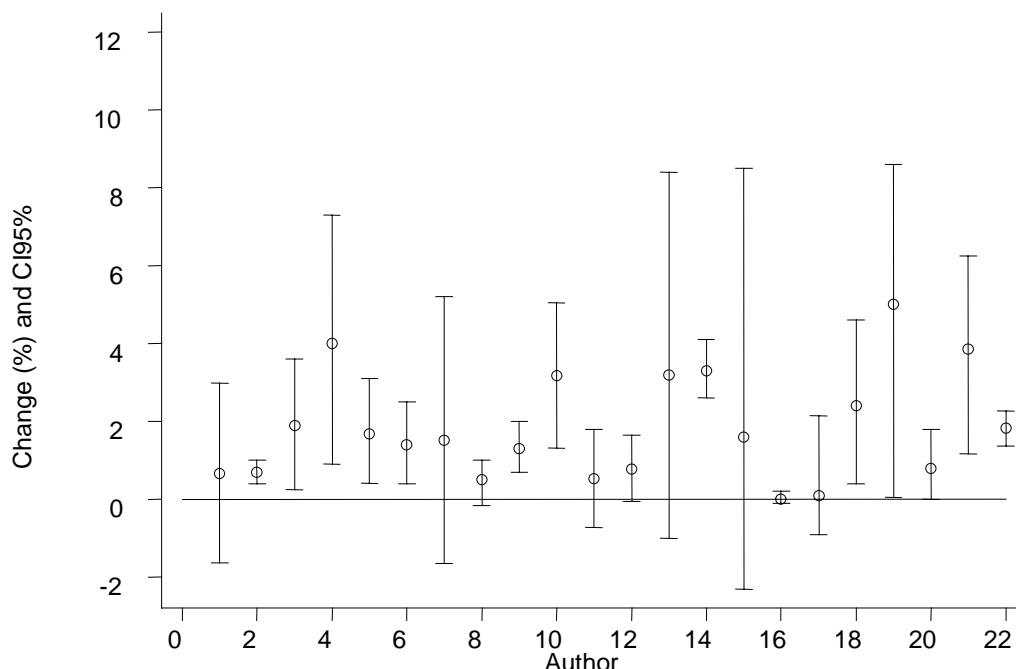
Note: The numbers represent the following studies: 1. Anderson 1996 (London), 2. Ballester 1996 (Valencia), 3. Borja-Aburto 1997 (Mexico), 4. Brenner 1999 (London), 5. Dockery 1992 (St. Louis), 6. Dockery 1992 (Tennessee), 7. Gamble 1996 (Chicago), 8. Gamble 1996 (Utah), 9. Ito 1996 (Chicago), 10. Kelsall 1997 (Philadelphia), 11. Kinney 1995 (Los Angeles), 12. Lee 1999 (Seoul), 13. Mazundar 1983 (Pittsburgh), 14. Moolgavkar 1996<sup>b</sup> (Ohio), 15. Moolgavkar 1996<sup>a</sup> (Philadelphia), 16. Neas 1999 (Philadelphia), 17. Ostro<sup>b</sup> 1995 (California), 18. Ostro 1996 (Santiago), 19. Pope III 1996 (Utah), 20. Pope III 1999 (Ogden), 21. Pope III 1999 (Provo), 22. Pope III 1999 (Utah), 23. Samet 1998 (Philadelphia), 24. Schwartz 1994<sup>c</sup> (Cincinnati), 25. Schwartz 1992<sup>a</sup> (Philadelphia), 26. Schwartz 1992<sup>b</sup> (Steubenville), 27. Schwartz 1993<sup>a</sup> (Birmingham), 28. Schwartz 1991 (Detroit), 29. Schwartz 1996<sup>b</sup> (Ohio), 30. Simpson 1997 (Brisbane), 31. Spix 1993 (Erfurt), 32. Sunyer 1996 (Barcelona), 33. Touloumi 1993 (Athens), 34. Touloumi 1996 (Athens), 35. Verhoeff 1996 (Amsterdam), 36. Wordley 1997 (Birmingham), 37. Zmirou 1996 (Lyon), 38. Castillejos 2000 (México), 39. Cropper 2000 (Delhi), 40. Pooled estimate.

Figure 1 shows the percent change in general mortality associated with an increase in air pollution. The percent change, considering all the cases, establishes an increase in mortality of between 0.06 and 2.82% with a weighted estimate of 1.01 (CI 95% 0.83-1.18). These data are for total, non-accidental deaths.

Despite the consistency of this association with excess mortality there are aspects of this association that are still uncertain. There is always concern that some confounder, another variable correlated with the exposure and causally related to the effect, might actually be responsible for an association found by an epidemiological study. However, many studies have separated the effects of particles including other pollutants in the statistical models. The coherence of associations with other effects makes this association plausible. Additionally, clinical studies have demonstrated decreased lung function, increased frequencies of respiratory symptoms, heightened airway hyper-responsiveness, and cellular and biochemical evidence of lung inflammation in exercising adults exposed to ozone concentrations at low exposures.

The pooled estimate we obtained is larger than that obtained by Levy (2000) because of the inclusion of more worldwide recent reports. Although the above results are significant, death could be more certainly attributed to air pollution exposure if the cause of death were determined as due to some ailment which is caused or aggravated by air pollution, such as death due to respiratory or cardiac diseases (Figures 2 and 3).

Figure 2. Percent change in mortality due to respiratory causes for each  $10 \mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$

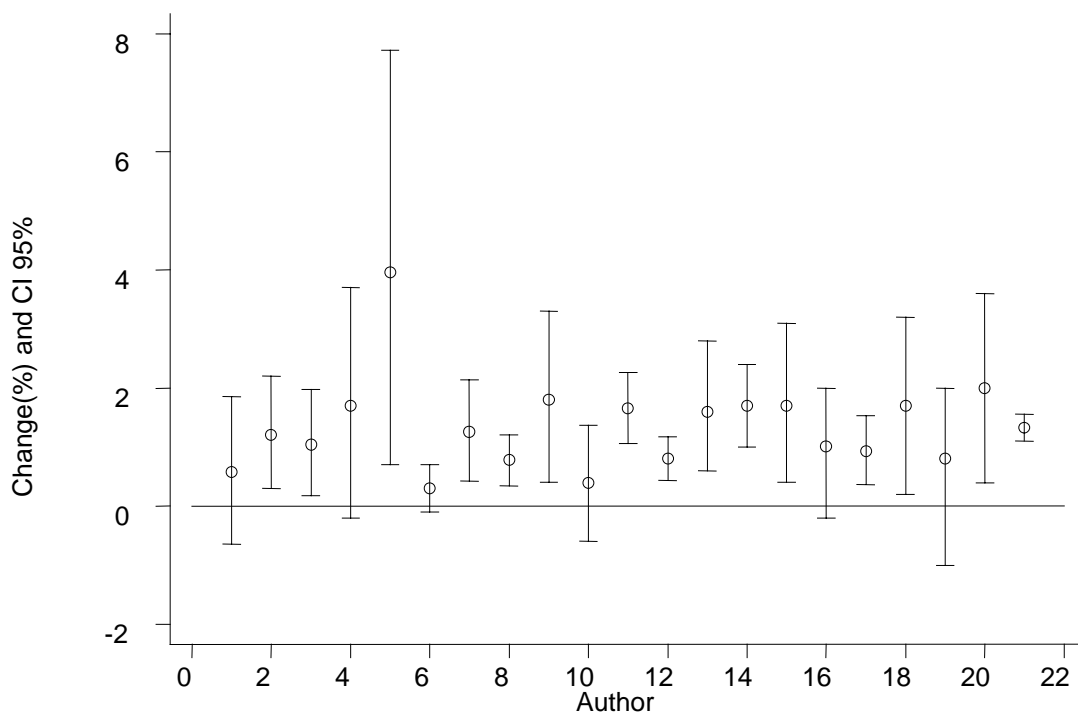


Note: The numbers represent the following studies: 1. Anderson 1996 (London), 2. Ballester 1996, (Valencia), 3. Borja-Aburto 1997 (Mexico), 4. Brenner 1999 (London), 5. Dab 1996 (París), 6. Ito 1996 (Chicago), 7. Neas 1999 (Philadelphia), 8. Ostro 1995<sup>b</sup> (California), 9. Ostro 1996 (Santiago), 10. Pope III 1999 (Ogden), 11. Pope III 1999 (Provo), 12. Pope III 1999 (Utah), 13. Schwartz 1994<sup>c</sup> (Cincinnati), 14. Schwartz 1992<sup>b</sup> (Philadelphia), 15. Schwartz 1993<sup>a</sup> (Birmingham), 16. Simpson 1997 (Brisbane), 17. Sunyer 1996 (Barcelona), 18. Vigotti 1996 (Milan), 19. Wordley 1997 (Birmingham), 20. Zmirou 1996 (Lyon), 21. Castillejos 2000 (Mexico), 22. Pooled estimated.

Figure 2 shows the studies where an increase in death due to respiratory causes was evaluated with high levels of  $\text{PM}_{10}$  pollution. The increases are greater than those describing total death, with a range of percent increase from 0.4 to 5.0%. Only the two studies by Simpson *et al.* in 1997 (0.01%) and Sunyer *et al.* in 1996 (0.09%) reported low increases. For these studies the pooled estimated is greater than that reported for total, non-accidental death, 1.82 (CI 95% 1.37-2.22).

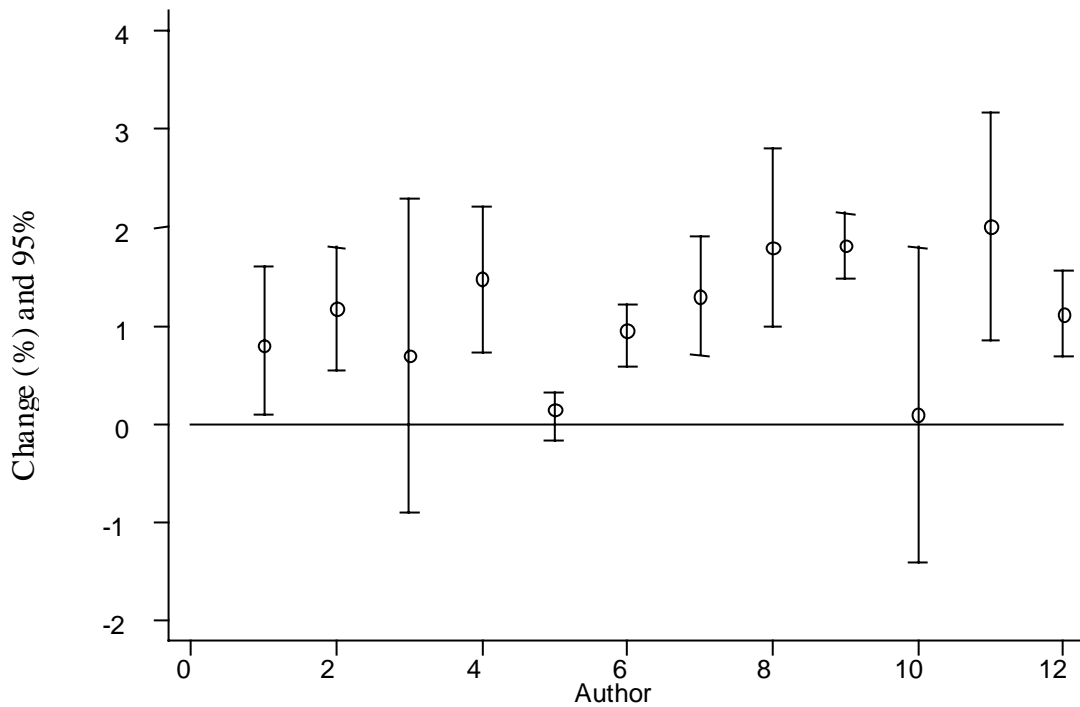
Studies that have determined an increase in death due to cardiovascular system damage associated with exposure to  $\text{PM}_{10}$  are summarised in Figure 3. In this case the range of percent change is lower than for deaths due to respiratory ailments (0.30 to 1.80%). Only the 1996 Gamble *et al.* study reported percentages above 3% (3.96%). The weighted average is 1.32 (CI 95% 1.10-1.55).

Figure 3. **Percent change in mortality due to cardiovascular causes for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**



Note: The numbers represent the following studies: 1. Anderson 1996 (London), 2. Ballester 1996 (Valencia), 3. Borja-Aburto 1997 (Mexico), 4. Bremner 1999 (London), 5. Gamble 1996 (Utah), 6. Ito 1996 (Chicago), 7. Neas 1999 (Philadelphia), 8. Ostro 1996 (Santiago), 9. Pope III 1996 (Utah), 10. Pope III 1999 (Ogden), 11. Pope III 1999 (Provo), 12. Pope III 1999 (Utah), 13. Schwartz 1994<sup>c</sup> (Cincinnati), 14. Schwartz 1992<sup>a</sup> (Philadelphia), 15. Schwartz 1993<sup>a</sup> (Birmingham), 16. Simpson 1997 (Brisbane), 17. Sunyer 1996 (Barcelona), 18. Wordley 1997 (Birmingham), 19. Zmirou 1996 (Lyon), 20. Castillejos 2000 (Mexico), 21. Pooled estimate.

Figure 4. **Percent change in mortality for individuals older than 65 years for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**



Note: The numbers represent the following studies: 1. Ballester 1996 (Valencia), 2. Borja-Aburto 1997 (Mexico), 3. Brenner 1999 (London), 4. Neas 1999 (Philadelphia), 5. Ostro 1995<sup>b</sup> (California), 6. Ostro 1996 (Santiago), 7. Saldiva 1994 (Sao Paulo), 8. Schwartz 1994<sup>c</sup> (Cincinnati), 9. Schwartz 1992<sup>a</sup> (Philadelphia), 10. Simpson 1997 (Brisbane), 11. Castillejos 2000 (Mexico), 12. Pooled estimate.

Once again, the elderly, those individuals 65 years of age and older, must be dealt with in an independent analysis from the rest of the population, because their physiology renders them at high risk of suffering toxic effects from exposure to air pollution. Figure 4 summarises the major studies where increases in total mortality (non-accidental) have been reported associated with exposure to  $\text{PM}_{10}$ . Percent change for these studies varies from 0.1 to 1.82%. The pooled estimate is 1.18 (CI 95% 0.66-1.57).

*b) Infant mortality associated with exposure to  $\text{PM}_{10}$*

Only a few studies document the association of infant mortality associated with  $\text{PM}_{10}$  is very important. To date only three publications report an increase in post neonatal mortality (Table 4). Two of these studies were performed in the U.S. and the other in the Czech Republic. The U.S. studies reported a percent change from 1.05 to 1.20%, while the Czech study showed an increase between 3.65 and 7.08%.

The results shown in Table 4, however, demonstrate differences in the magnitude of the changes in increased mortality. In the Czech study, the increase in mortality for respiratory disease associated deaths is almost twice the increase in general, non-accidental deaths. The Woodrouff *et al.* study, on the other hand, reports very slight differences. Despite the differences, both studies indicate increased death. Another interesting result from these studies is that low birth weight babies (1.05%) had a lesser increase in mortality than normal birth weight babies (1.20%). The studies also reported that deaths from sudden infant death syndrome increased more (1.12%) than deaths from other causes (1.04%).

Two studies have reported on neonate and infant death associate with exposure to PM<sub>10</sub>. One study was carried out in the Czech Republic (cross-sectional) and the other in Mexico (time-series)

General, non-accidental mortality was reported as more than twice as high for infants as for neonates. Such a difference could be due to greater exposure for infants than neonates.

The relationship between parental exposure to high concentrations of PM<sub>10</sub> and low birth weight is another relevant toxicological parameter. To date only one study by Wang *et al.*, 1998, has dealt with this topic. Wang reported on infants born between 1988 and 1991 with a significant 1% decrease in new-born weight associated with mothers exposures to PM<sub>10</sub> concentrations between 9 and 308 µg/m<sup>3</sup> (CI 95% 0.5-1.4).

Table 4. **Percent change in mortality post neonatal, neonate and infant for each 10 µg/m<sup>3</sup> increase in PM<sub>10</sub>**

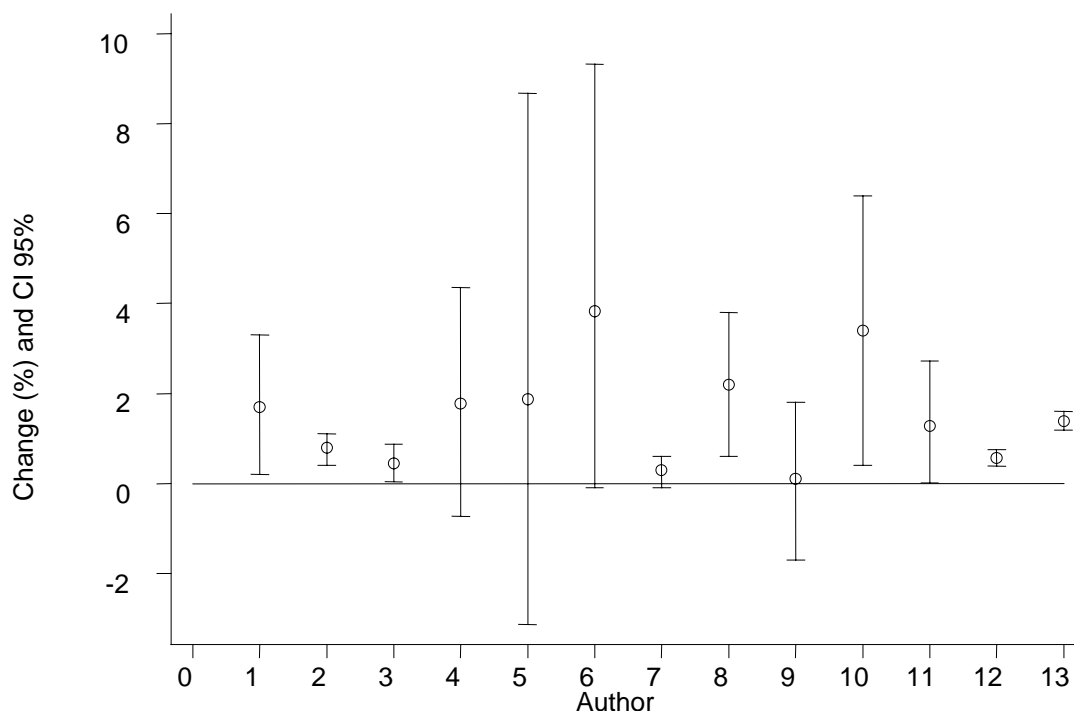
Author	Year	Effects	Study	Country	Period	% change	95% CI	
							LL	UL
Bobak	1992	• Post neonatal mortality	Cross-sectional study	Czech Rep.	1986 to 1988	3.65	0.59	7.43
Bobak	1992	• Death associated with respiratory system	Cross-sectional study	Czech Rep.	1986 to 1988	7.08	4.25	47.93
Woodroff	1997	• Post neonatal mortality	Cross-sectional study	U.S.A.	1989 to 1991	1.04	1.02	1.07
Woodroff	1997	• Death associated with respiratory system • Sudden infant death syndrome • Normal birth weight	Cross-sectional study	U.S.A.	1989 to 1991	1.12	1.07	1.17
Woodroff	1997	• Death associated with respiratory system • Normal birth weight	Cross-sectional study	U.S.A.	1989 to 1991	1.20	1.06	1.36
Woodroff	1997	• Death associated with respiratory system • Low birth weight	Cross-sectional study	U.S.A.	1989 to 1991	1.05	0.91	1.22
Bobak	1992	• Infant and neonate mortality	Cross-sectional study	Czech Rep.	1986 to 1988	1.65	-0.23	3.77
Loomis	1999	• Infant and neonate mortality	Time-series study	Mexico DF	1993 to 1995	3.52	0.72	6.31

c) *Percent change in hospitalisations due to respiratory diseases*

The number of individuals hospitalised due to respiratory ailments for a given period is another useful indicator often employed to determine the effects of exposure to low concentrations of air pollution (specifically  $PM_{10}$ ) on the population.

Figure 5 shows some of the studies where an association has been assessed between pollution levels and increased hospitalisations due to respiratory causes. The reported increases adjusted to a change of 10 units of pollutant concentration were between 0.30 and 3.83%. All of these studies were carried out exclusively in developed countries, which points out the need for the same type of research in developing nations, where exposure to environmental pollutants tends to be greater. In this case the pooled estimate increase was 1.39 (CI 95% 1.18-1.60).

Figure 5. **Percent change in hospitalisations due to respiratory ailments for 10  $\mu\text{g}/\text{m}^3$  increase in  $PM_{10}$**

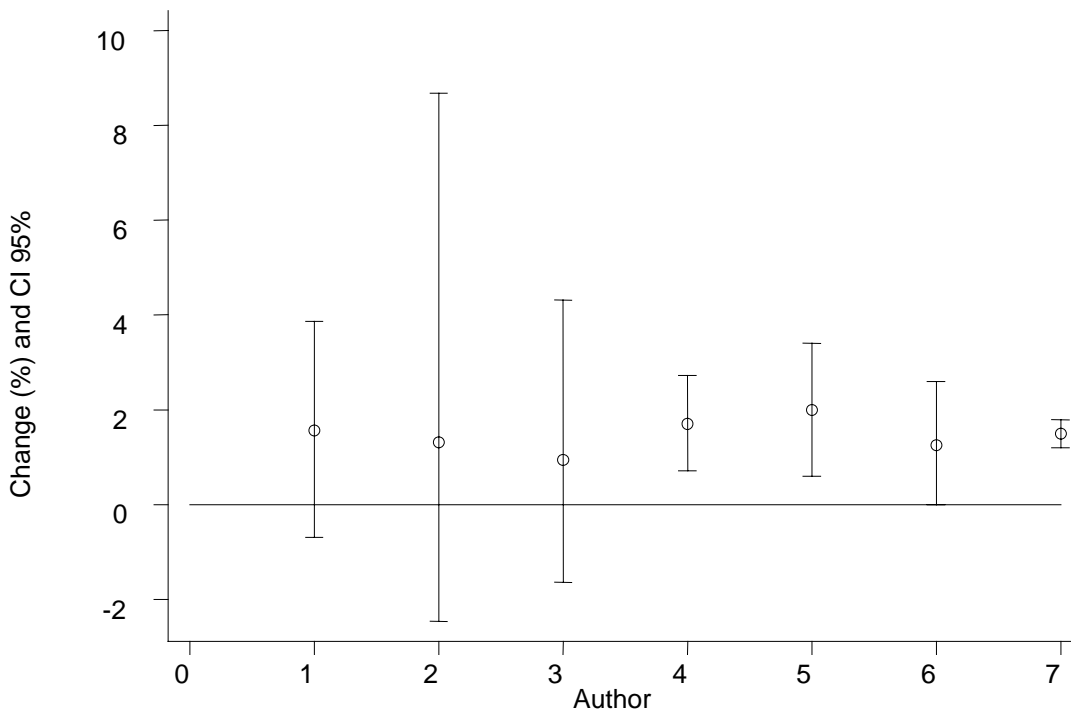


Note: The numbers represent the following studies: 1. Abbey 1995 (California), 2. Burnett 1995 (Ontario), 3. Dab 1996 (Paris), 4. Ponce de León 1996 (London), 5. Schouten 1996 (Amsterdam), 6. Schouten 1996 (Rotterdam), 7. Schwartz 1996<sup>a</sup> (Cleveland), 8. Thurston 1992 (Buffalo), 9. Thurston 1992 (New York), 10. Thurston 1994 (Ontario), 11. Vigotti 1996 (Milan), 12. Linn 2000 (Los Angeles). 13. Pooled estimate.



Figure 6 summarises the studies where an association was established between hospitalisations due to respiratory ailments and exposure in individuals older than 65 years of age. The trend of the changes was the same with an average increase ranging between 0.94 and 1.70% . The weighted average is 1.49 (CI 95% 1.20 – 1.78).

Figure 6. **Percent change in hospitalisations due to respiratory diseases in individuals older than 65 years for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**

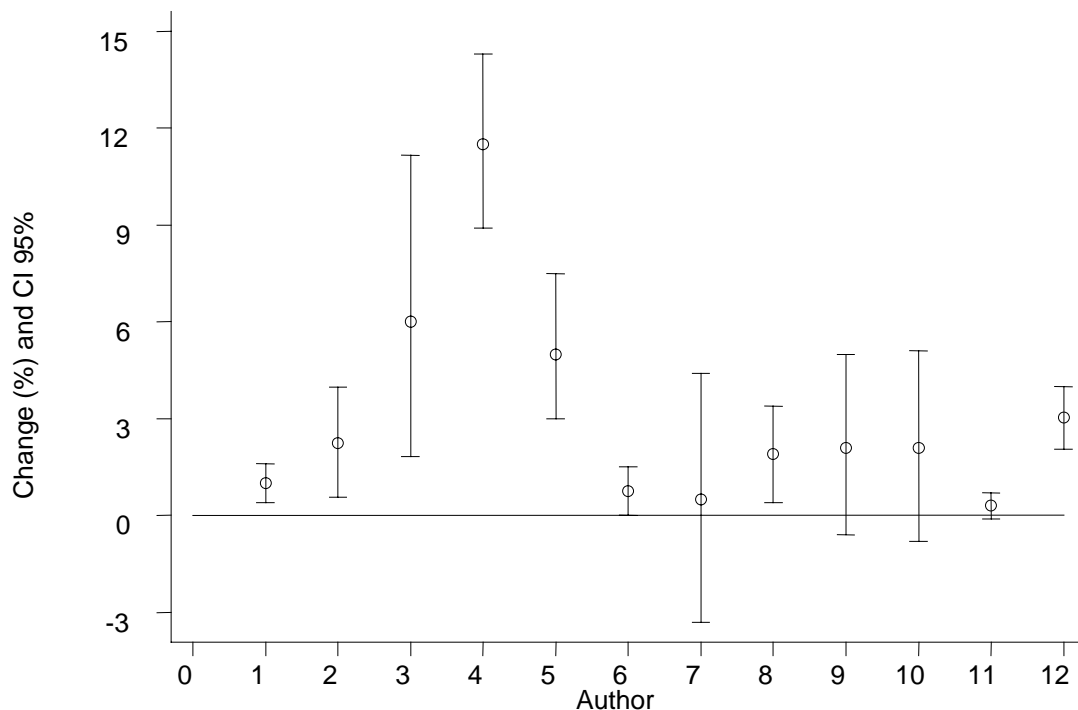


Note: The numbers represent the following studies: 1. Ponce de León 1996 (London), 2. Schouten 1996 (Amsterdam), 3. Schouten 1996 (Rotterdam), 4. Schwartz 1999 (Spokane), 5. Schwartz 1999 (New Heaven), 6. Schwartz 1995<sup>a</sup> (Tacoma), 7. Pooled estimate.

Besides establishing the increase in hospitalisations due to  $\text{PM}_{10}$ , the types of diseases for which patients were hospitalised and are most associated with exposure should also be determined in order to recognise which individuals will be more at risk during an episode of elevated environmental pollution. Figures 7-9 show the association between  $\text{PM}_{10}$  exposure and hospitalisation for asthma, COPD (chronic obstructive pulmonary disease) and pneumonia, respectively.

Figure 7 summarises studies where an association was found between PM<sub>10</sub> levels and hospitalisation for asthma attacks. The results from the different studies show a general increase in the percent change from 0.5 to 11.5%. The pooled estimate increase was 3.02% (CI 95% 2.05 - 4.00).

Figure 7. **Percent change in hospitalisations for asthma for each 10 µg/m<sup>3</sup> increase in PM<sub>10</sub>**

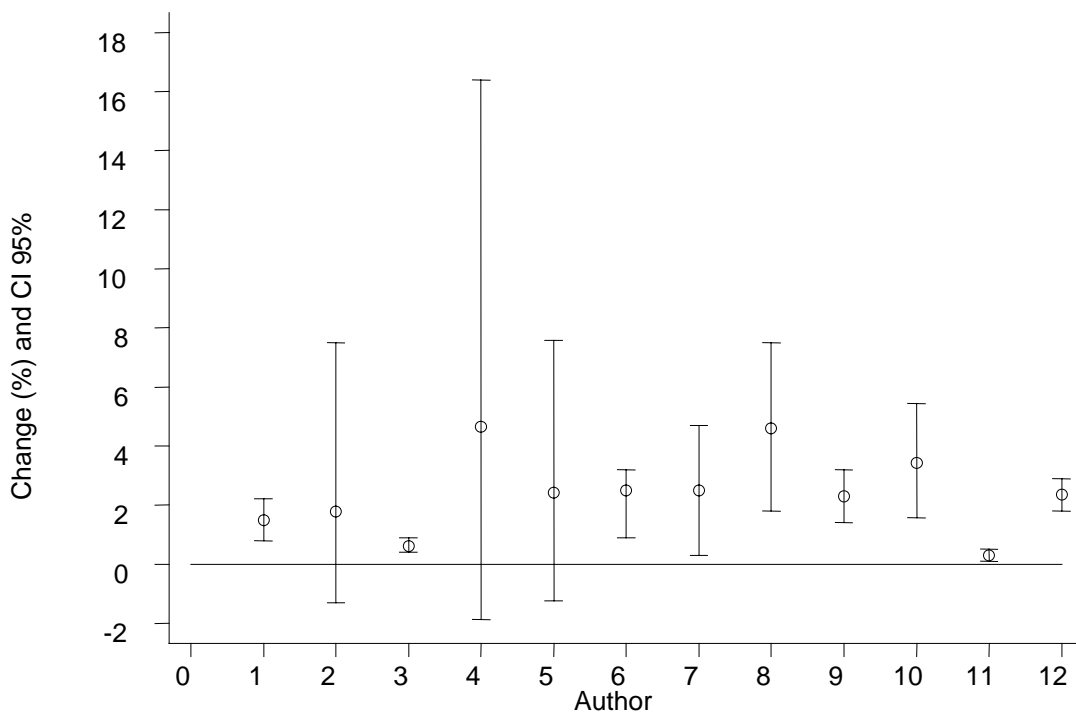


Note: The numbers represent the following studies: 1. Burnett 1995 (Ontario), 2. Delfino 1994 (Montreal), 3. Lipsset 1997 (Santa Clara), 4. Ostro 1991 (Denver), 5. Romieu 1996 (Mexico), 6. Sheppard 1999 (Seattle), 7. Schwartz 1995<sup>a</sup> (Tacoma), 8. Thurston 1992 (Buffalo), 9. Thurston 1992 (New York), 10. Thurston 1994 (Ontario), 11. Linn 2000 (Los Angeles), 12. Pooled estimate.

Figure 8 summarises the studies where a significant association was established between increased hospitalisation for COPD and PM<sub>10</sub> levels in the general population. The percent change ranged from 0.6 to 4.66%. The pooled estimated increase was 2.34% (CI 95% 1.80 - 2.89).

In this same category, the percent increase for individuals older than 65 years of age was clearly higher than for the rest of the population (Schwartz, 1999) (not shown in the figure).

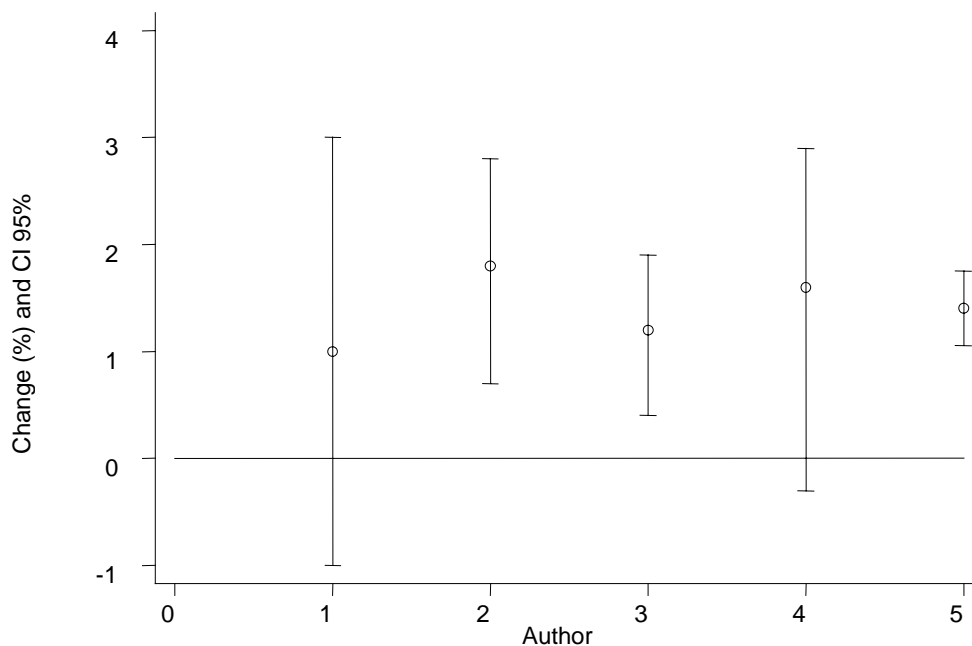
Figure 8. **Percent change in hospitalisations due to COPD for each 10 µg/m<sup>3</sup> increase in PM<sub>10</sub>**



Note: The numbers represent the following studies: 1. Burnett 1995 (Ontario), 2. Moolgavcar 1997 (Birmingham), 3. Ostro 1995 (Santiago), 4. Schouten 1996 (Amsterdam), 5. Schouten 1996 (Rotterdam), 6. Schwartz 1993<sup>a</sup> (Birmingham), 7. Schwartz 1995<sup>a</sup> (Tacoma), 8. Schwartz 1994<sup>d</sup> (Minneapolis), 9. Sunyer 1995 (Barcelona), 10. Schwartz 1999 (Spokane), 11. Linn 2000 (Los Angeles) 12. Pooled estimate.

Pneumonia is another disease of the pulmonary system for which increased incidence has been reported associated to exposure to  $PM_{10}$ . Figure 9 shows the major studies realised to date on this topic. All the studies were carried out in the U.S. and published by Schwartz *et al.* and Moolgavcar *et al.* For pneumonia, the increases reported ranged from 1.2 to 1.8%. The pooled estimated increase 1.40% CI 95% 1.05 - 1.75) was greater than for COPD.

Figure 9. Percent change in hospitalisations due to pneumonia for each  $10 \mu\text{g}/\text{m}^3$  increase in  $PM_{10}$

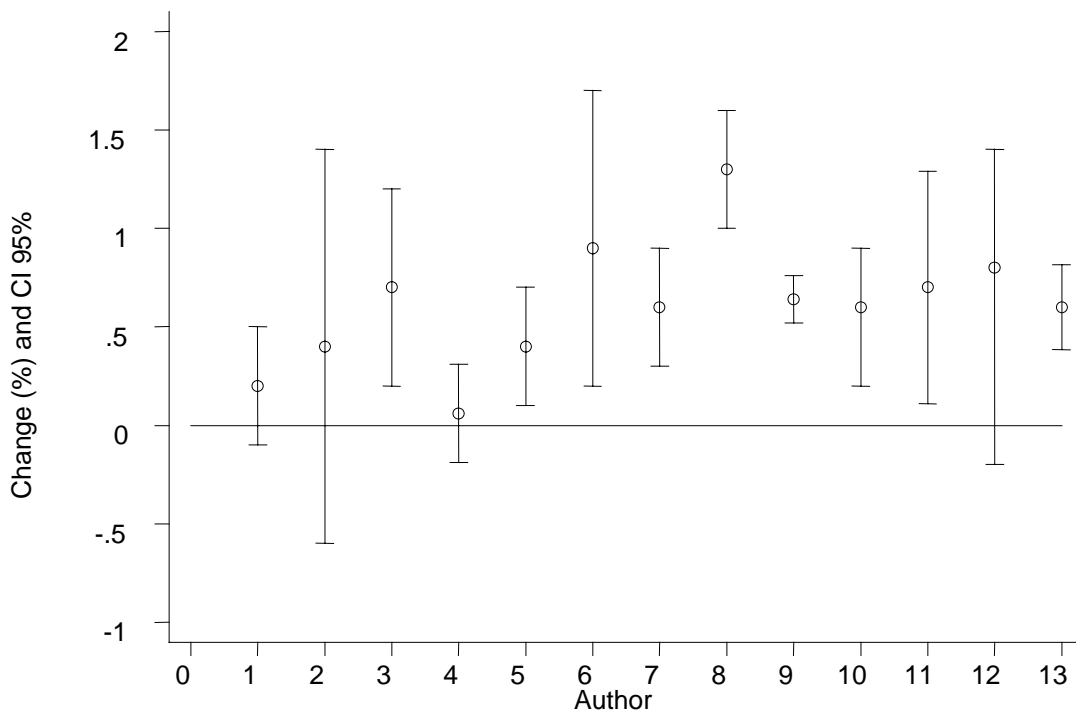


Note: The numbers represent the following studies: 1. Moolgavkar 1997 (Minneapolis), 2. Schwartz 1993<sup>a</sup> (Birmingham), 3. Schwartz 1995<sup>a</sup> (Tacoma), 4. Schwartz 1994<sup>d</sup> (Minneapolis), 5. Pooled estimate.

It is logical that respiratory diseases should be used as a first choice parameter in determining adverse effects associated with air pollution. Hospitalisation for cardiac ailments is also an important parameter for determining harmful exposure to  $PM_{10}$ . Figures 10 and 11 summarise these studies.

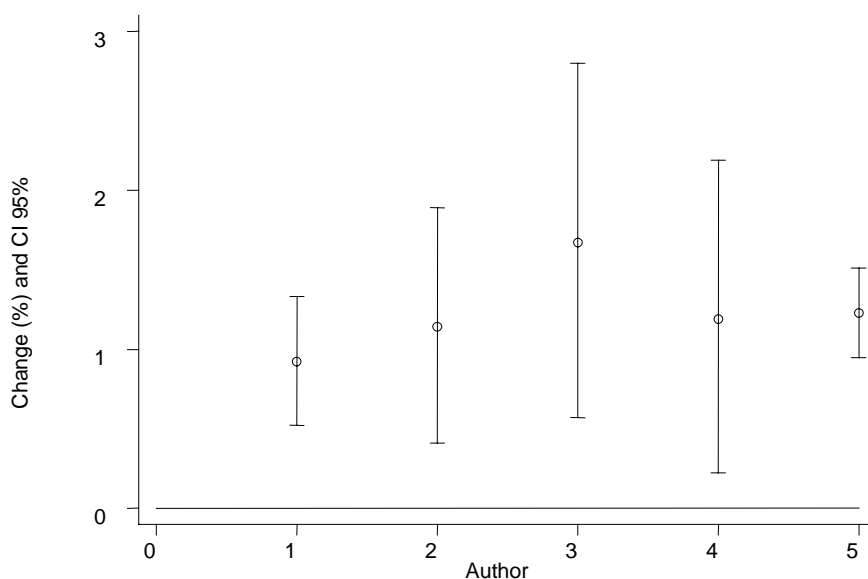
Figure 10 shows the percent change in hospitalisations due to cardiac diseases in all ages with increases ranging from 0.40 to 0.90%. Weighted average 0.60% (CI 95% 0.42 - 0.79). Figure 11 shows the effect for individuals older than 65 years of age with all increases above 1.22% (95% CI 0.94 - 1.50). All percent changes and the pooled estimated increase (1.22 vs. 0.60) are greater than those in Figure 10. This reiterates the importance of considering this age group specifically.

Figure 10. **Percent change in hospitalisations due to cardiovascular diseases for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**



Note: The numbers represent the following studies: 1. Linn 2000 (Los Angeles) arrhythmia), 2. Burnett 1995 (Ontario) (arrhythmia), 3. Burnett 1995 (Ontario) (coronary artery), 4. Linn 2000 (Los Angeles) (cerebrovascular), 5. Linn 2000 (Los Angeles) (congestive hearth failure), 6. Burnett 1995 (Ontario) (congestive hearth failure), 7. Linn 2000 (Los Angeles) (myocardial infarction), 8. Linn 2000 (Los Angeles) (occlusive stroke), 9. Linn 2000 (Los Angeles) (total), 10. Schwartz 1995<sup>b</sup> (Detroit) (total), 11. Schwartz 1997 (Michigan) (total), 12. Morris 1998 (Chicago) (total), 13. Pooled estimate.

Figure 11. **Percent change in hospitalisations for cardiac disease in individuals more than 65 years old for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**



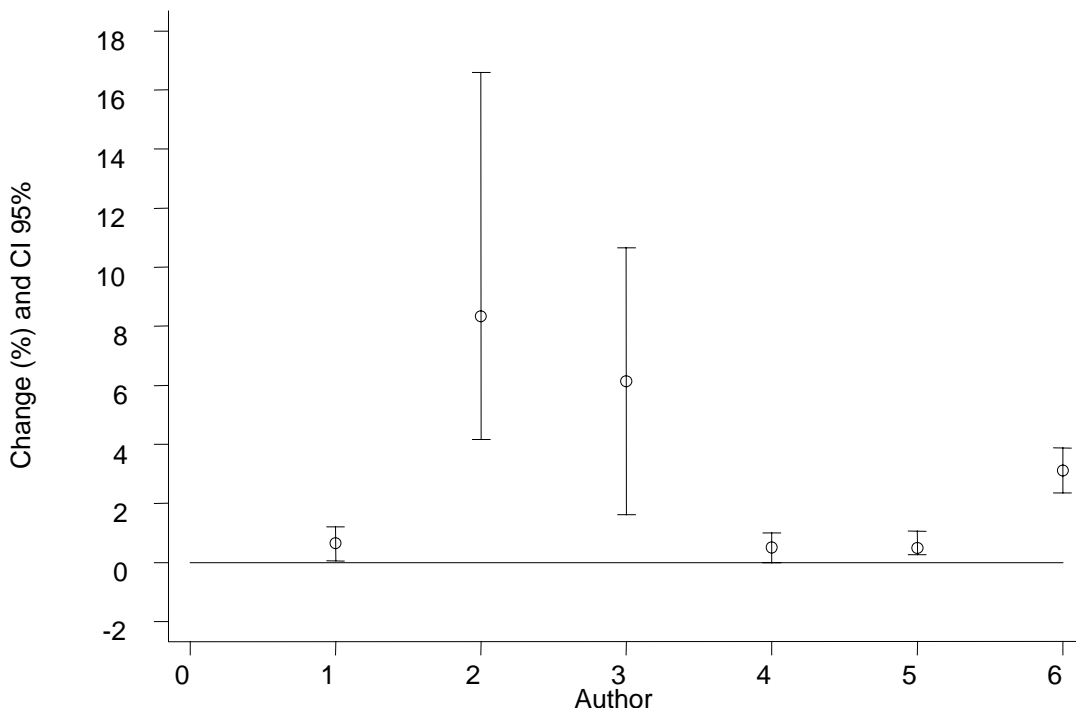
Note: The numbers represent the following studies: 1. Schwartz 1999 (Chicago), 2. Schwartz 1999 (New Heaven), 3. Schwartz 1999 (St Paul), 4. Schwartz 1997 (Tucson), 5. Pooled estimate.

d) *Percent change in hospital emergency room admissions*

Hospitalisations are not the only parameter useful for chronic exposure studies of  $\text{PM}_{10}$ . Hospital emergency room admissions for respiratory ailments are also considered as an indicator. The number of studies analysing this factor is much lower than for hospitalisation studies, probably due to the lack of complete and accurate records for these patients.

Figure 12 summarises some of the studies that report an association between increased emergency room admissions due to respiratory ailments and increased pollutant concentration. The increases determined vary widely up to 8.34% with a pooled estimated increase of 3.11% (CI 95% 2.35 - 3.88).

Figure 12. **Percent change in hospital emergency room admissions due to respiratory causes for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**



Note: The numbers represent the following studies: 1. Atkinson 1999 (London), 2. Damakosh 2000 (Mexico) (low respiratory symptoms), 3. Delfino 1997 (Quebec), 4. Samet 1995 (Steubenville), 5. Atkinson 1999 (London), 6. Pooled estimate.

The increased emergency room admissions for children's asthma attack associated with exposure to particles was 4.50% (CI 95% 2.16 - 7.0) in a study by Lipset (for a childhood study).

Increased emergency room admissions associated with increased pollutant levels have been evaluated for other conditions as well with the reported results for croup (2.48%), tracheitis (12.5%), pneumonia (20.8%) and total admission (3.40%). Pneumonia shows an especially high increase in emergency room treatment (Table 5).

Table 5. **Percent change in hospital emergency room admission for different respiratory ailments for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**

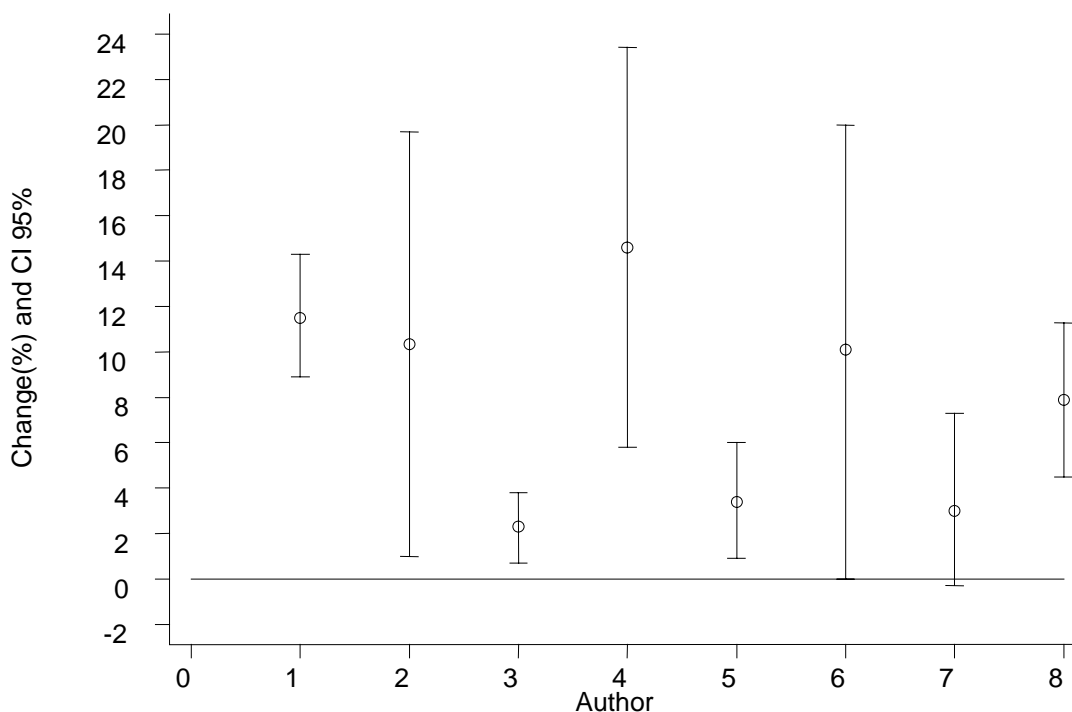
Author	Year	City	Period	PM <sub>10</sub> levels	% change	95% CI	
				Mean (range)		LL	UL
Schwartz <sup>1</sup>	1993 <sup>c</sup>	Seattle	1989 – 1990	26.9 (6.00 - 103.0)	3.40	0.90 to 6.00	
Damakosh <sup>2</sup>	2000	Mexico	1993 – 1994	45.0 ( 48.0 - 121.0)	12.5	0.00 to 29.2	
Damakosh <sup>3</sup>	2000	Mexico	1993 – 1994	45.0 ( 48.0 - 121.0)	20.8	4.16 to 45.8	
Schwartz	1991	Germany	1983 - 1985	32.40 ( 16.8 – 70.20)	2.48	3.10 to 4.34	

<sup>1</sup> Total visits, <sup>2</sup> Visits for tracheitis, <sup>3</sup> Visits for pneumonia, <sup>4</sup> Cruop.

e) *Percent change in different respiratory symptoms in asthmatic individuals*

Because individuals who suffer from asthma are especially susceptible to the effects of pollution, it is important to evaluate this population in detail. Figure 13 shows the results of several studies where an association was assessed between exposure and increased occurrence of asthmatic attacks. The reported increases range from 2.23% to 14.6%. Weighted average 7.87% (CI 95% 4.48 - 11.27).

Figure 13. **Percent change in the occurrence of asthma attacks for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**

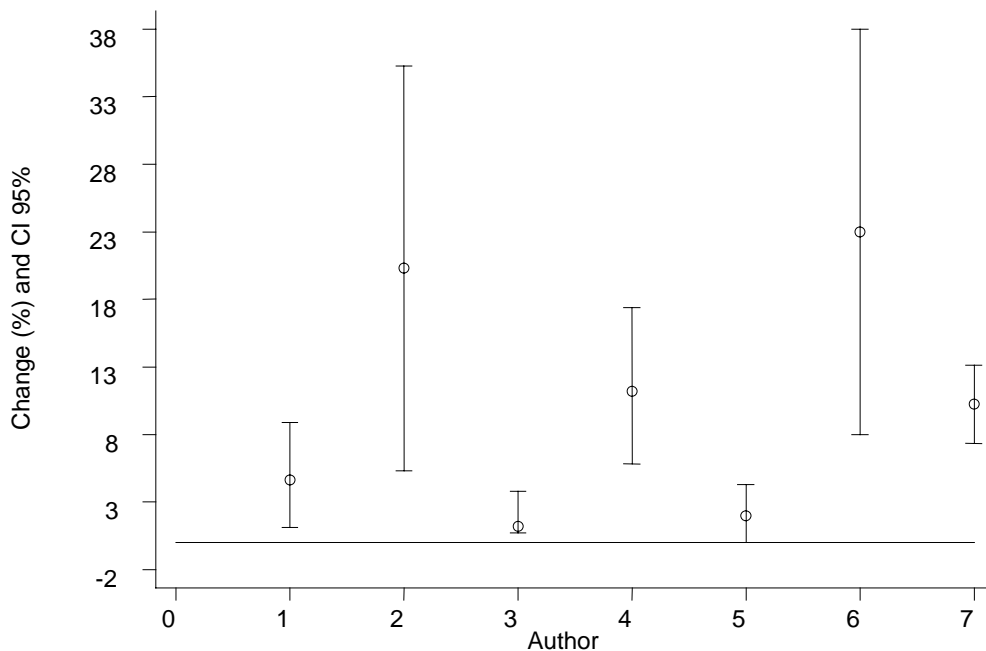


Note: The numbers represent the following studies: 1. Ostro 1991 (Denver), 2. Ostro 1995<sup>a</sup> (Los Angeles), 3. Roemer 1993 (Holland), 4. Neukrich 1998 (Paris) (adult), 5. Schwartz 1993<sup>c</sup> (Seattle), 6. Roemer 1993 (Holland) 7. Abbey 1995 (California) 8. Pooled estimate.



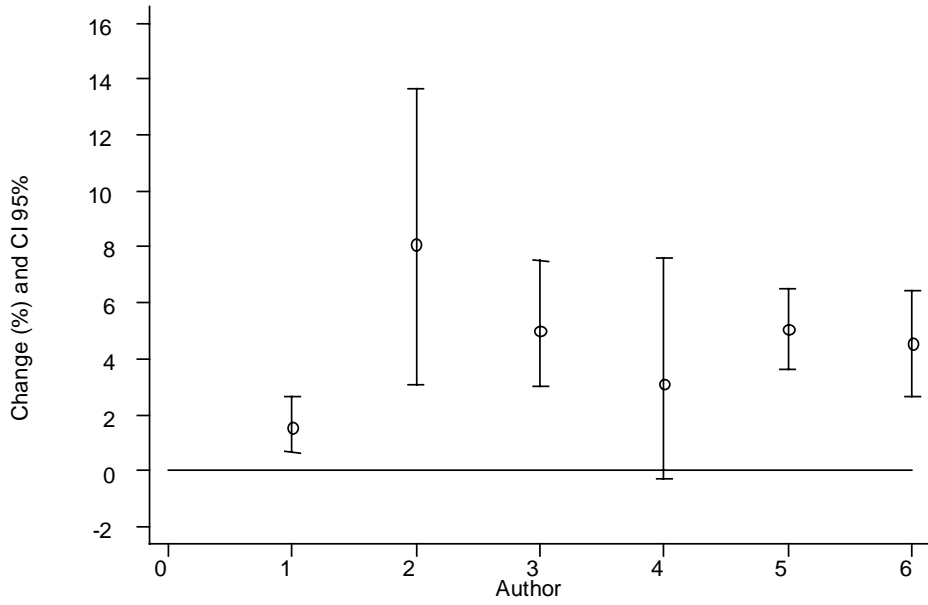
A closer look at this population, however, reveals that more severe effects are found for individuals who are undergoing medical treatment for their condition. It is possible that asthmatic symptoms are more severe in this group making them even more sensitive than others to the presence of pollutants. Percent changes for this type of study ranged from 4.48% (only one report showed an increase below 10%) to 20%, again with the greatest percent change appearing in a survey of adults. The pooled estimated for this group were similar than pooled estimates for the previous table.

Figure 14. **Percent change in the occurrence of asthma attacks and the use of bronchial dilators for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$  in children**



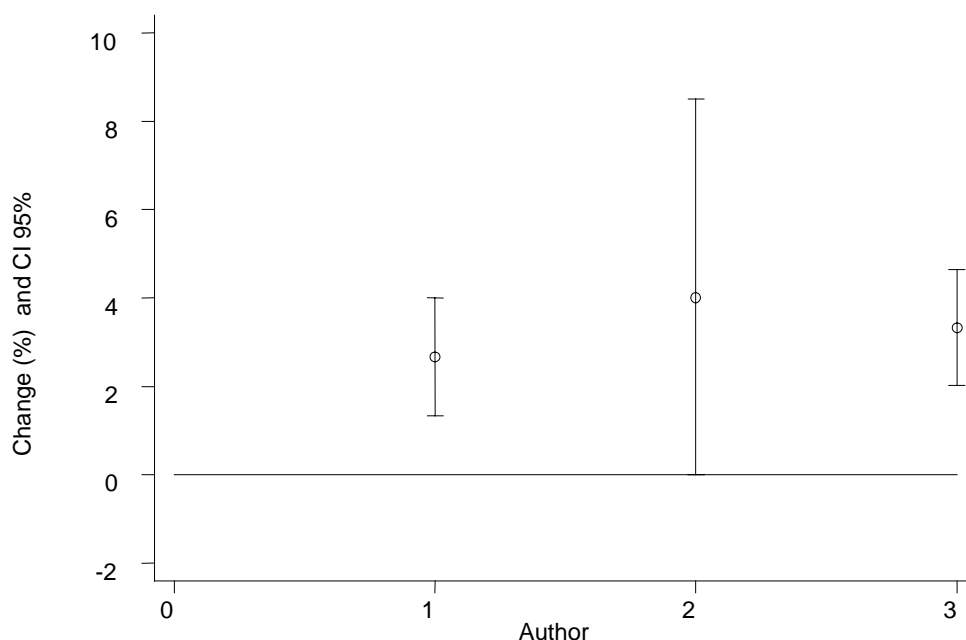
Note: The numbers represent the following studies: 1. Gielen 1997 (Amsterdam), 2. Peters 1997<sup>a</sup> (Sokolov), 3. Pope 1991<sup>a</sup> (Utah) (school population), 4. Pope 1991<sup>a</sup> (Utah), 5. Dusseldorp 1995 (Holland), 6. Romer 1993 (Holland), 7. Pooled estimate.

Figure 15. **Percent change in the presence of cough without phlegm in asthmatic children for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**



Note: The numbers represent the following studies: 1. Peters 1997<sup>a</sup> (Sokolov.), 2. Peters 1997<sup>b</sup> (Sokolov), 3. Romieu 1996 (Mexico), 4. Dusseldorp 1995 (Holland), 5. Pope 1992 (Utah), 6. Pooled estimate.

Figure 16. **Percent change in the presence of cough with phlegm in asthmatic children for  $10 \mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**



Note: The numbers represent the following studies: 1. Peters 1997<sup>a</sup> (Sokolov.), 2. Romieu 1996 (Mexico), 3. Pooled estimate.

Besides counting asthmatic attacks, the presence of a cough for asthmatics has also provided valuable results as a parameter for determining pollutant effects on the asthmatic population. The results of this type of study are summarised in Figures 15 and 16. In all cases, the results varied even for a single cough type. Increases reported for cough without phlegm ranged from 2.65% to 6.44% and for cough with phlegm, from 2.01 to 4.64%. These data again reaffirm the importance of this factor for susceptibility to environmental pollutants.

Table 6. **Percent change in the presence of different respiratory symptoms in asthmatic for each  $10 \mu\text{g}/\text{m}^3$   $\text{PM}_{10}$**

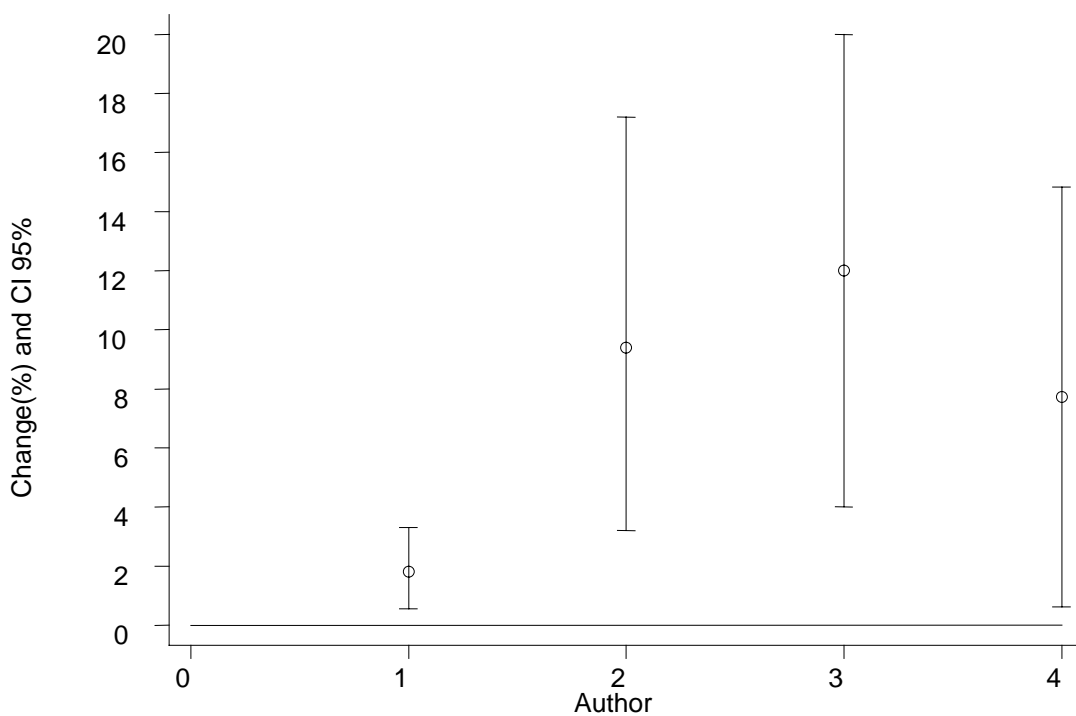
Author	Year	City	Period	$\text{PM}_{10}$ Levels Mean (range)	% change	95% CI	
						LL	UL
Peters <sup>1</sup>	1997 <sup>a</sup>	Sokolov	1991 – 1992	N.R.	24.60	1.62	59.23
Peters <sup>2</sup>	1997 <sup>b</sup>	Sokolov	1991 – 1992	47.0 (3.00 - 47.00)	1.33	0.44	2.22
Roemer <sup>2</sup>	1993	Holland	1990-1991	NR (2.00 – 120.0)	10.64	1.44	19.84
Dusseldorp <sup>2</sup>	1995	Holland	1993	NR	2.5	-2.10	9.70

<sup>1</sup> Fever. <sup>2</sup> Wheezing, NR= Not Reported. Peters's study was done on children.

f) *Percent change in different respiratory symptoms in the general population*

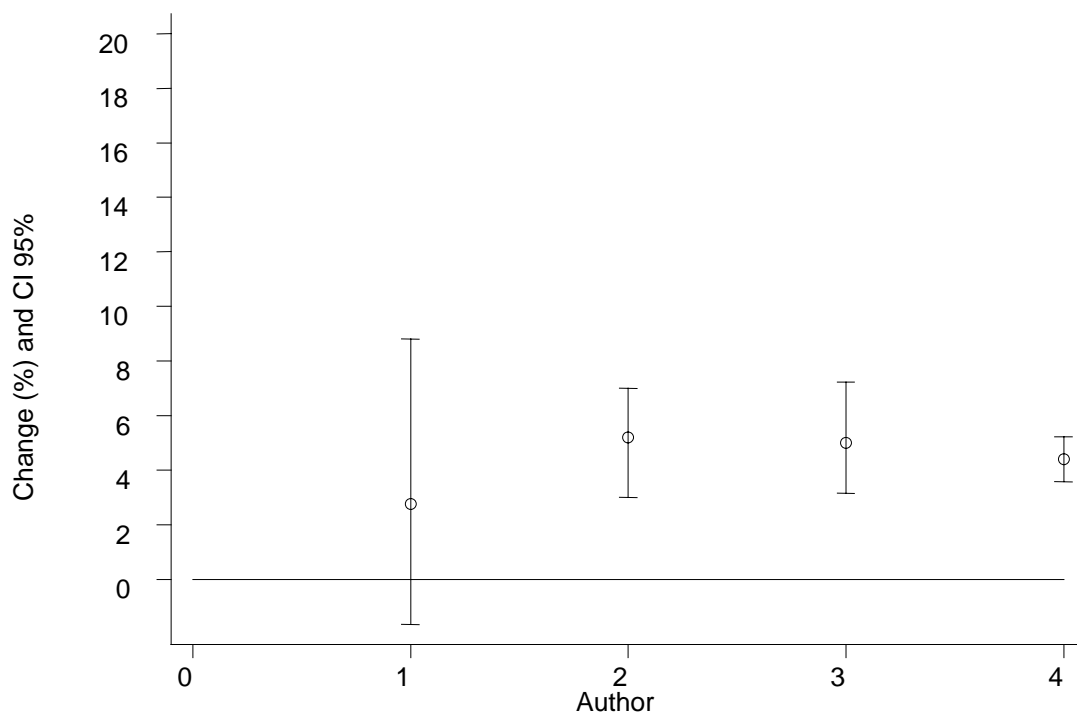
Although evaluating increased symptoms within populations whose age or health make them more susceptible than others to the toxic properties of pollutants is important, it is also crucial to study the effects on the rest of the population. Figure 17 of this section summarised the results obtained by associating the presence of respiratory symptoms with pollution levels within the general population. The reported increases vary from 1.8% to 12.0, with a weighted average of 7.72 (CI 95% 0.61 - 14.84).

Figure 17. **Percent change in the presence of respiratory symptoms in the general population for each  $10 \mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**



Note: The numbers represent the following studies: 1. Abbey 1993 (U.S.A.), 2. Peters 1997<sup>c</sup> (Erfurt), 3. Schwartz 1993<sup>b</sup> (U.S.A.), 4. Pooled estimated.

Figure 18. **Percent change in the presence of respiratory symptoms in the upper respiratory tract for each  $10 \mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**

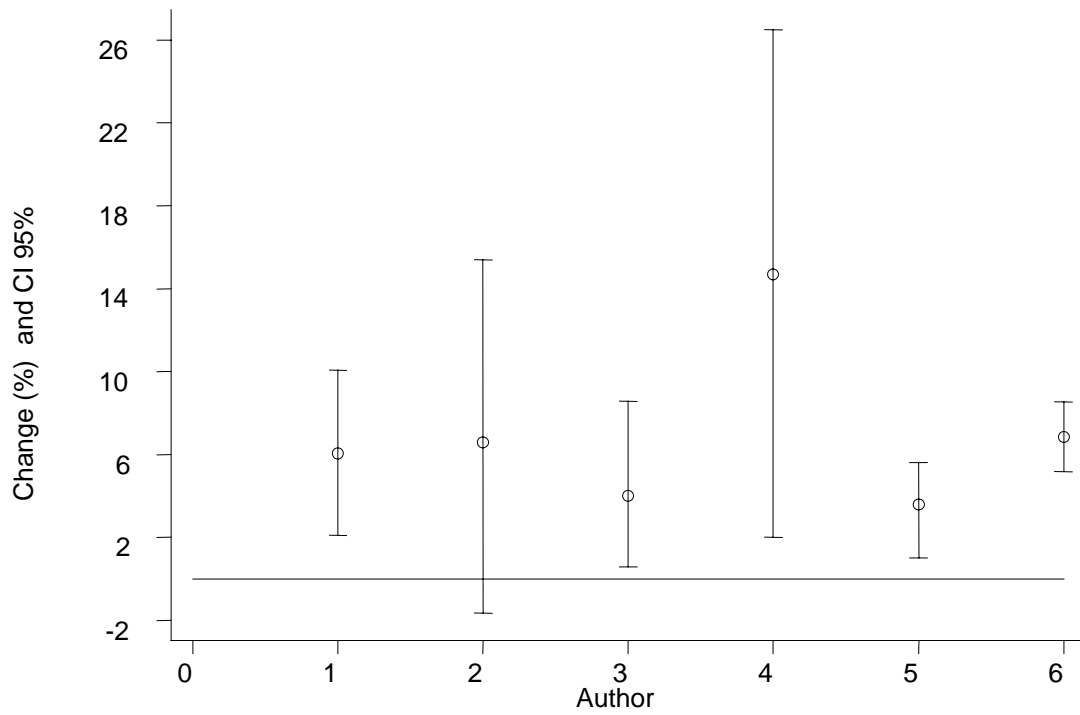


Note: The numbers represent the following studies 1. Ostro 1993 (California), 2. Pope 1991<sup>b</sup> (Utah), 3. Pope 1992 (Utah), 4. Pooled estimate.

Figure 18 summarises increases in symptoms specific to the upper respiratory tract. In the two studies which were carried out between 1989 and 1991, very similar increases (5.00 and 5.19%) were reported, while the lowest increase was reported for a study performed at the end of the 1970's (2.75%). The pooled estimate is 4.39 (CI 95% 3.56 - 5.12).

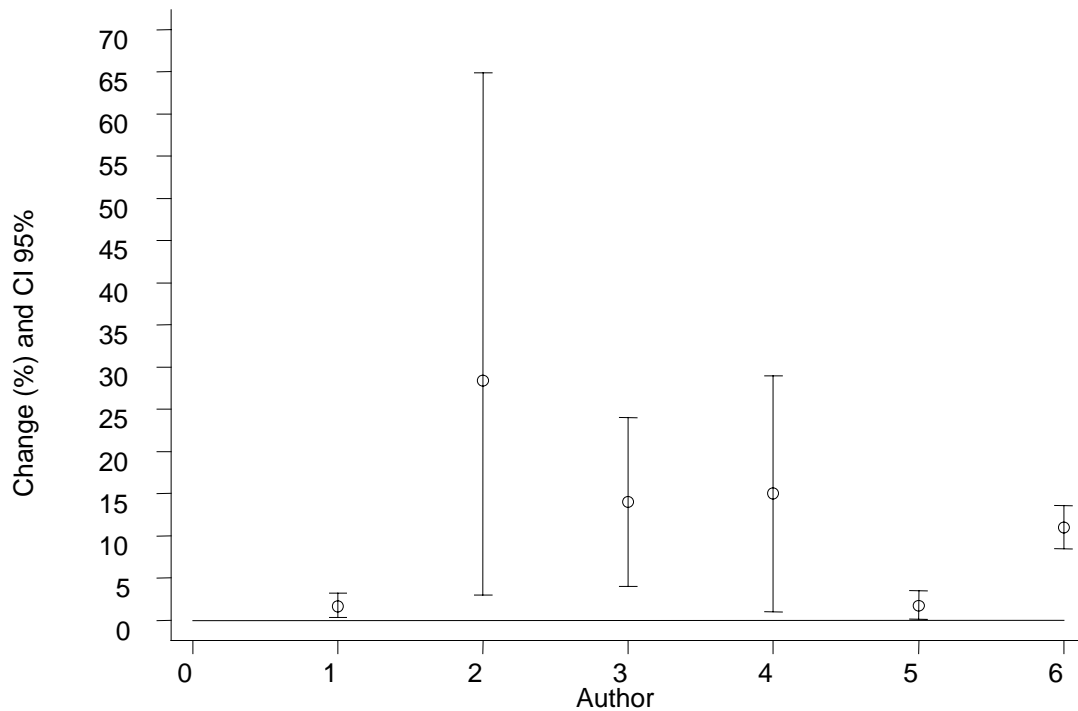
Changes in the presence of lower respiratory symptoms varied only slightly between 5% and 8.55% and are quite similar to those found in the previous figure. The largest increase of 14.7% considered corresponds to a study carried out on children. The pooled estimated 6.85%, (CI 95% 5.16 - 8.54) is greater than that for the previous figure (4.39%).

Figure 19. **Percent change in the presence of lower respiratory symptoms for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**



Note: The numbers represent the following studies: 1. Ostro 1993 (California), 2. Pope 1991<sup>b</sup> (Utah), 3. Pope 1992 (Utah), 4. Romieu 1996 (Mexico), 5. Gielen 1997 (Amsterdam), 6. Pooled estimate.

Figure 20. **Percent change in the presence of chronic bronchitis for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**



Note: The numbers represent the following studies: 1. Abbey 1993 (U.S.A.), 2. Aunan 1996 (U.S.A.), 3. Dockery 1989 (U.S.A.), 4. Schwartz 1993 (U.S.A.), 5. Abbey 1995 (California), 6. Pooled estimate.

Chronic bronchitis can be another useful parameter in determining the effects of exposure to  $\text{PM}_{10}$ . However, relatively few studies are available assessing the role of  $\text{PM}_{10}$  related to this ailment. Figure 20 shows four studies, all performed in the U.S. which found an increase in the presence of bronchitis associated with  $\text{PM}_{10}$  levels. Only one study by Abbey *et al.* reported a low increase of 1.65%.

Other respiratory symptoms have also been associated with exposure to pollutants as discussed above. Among these are the presence of a cough, shortness of breath and difficulty breathing upon awakening. In Table 7, the most significant increases from 6% to 27% were observed for the presence of a cough, followed by breathing difficulties upon awakening (4.8%) and shortness of breath (3.4%).

Table 7. **Percent change in the presence of different respiratory symptoms for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**

Author	Year	City	Period	PM <sub>10</sub> levels	% change	95% CI	
				Mean (range)		LL	UL
Dockery <sup>1</sup>	1989	U.S.A.	1980 - 1981	20.1 ( NR )	27.0	0.0 to 54.0	
Peters <sup>1</sup>	1997 <sup>c</sup>	Erfurt	1991 - 1992	60.0 (20.0 – 155.0)	6.00	1.8 to 11.0	
Zemp <sup>1</sup>	1999	Switzerland	1991	21.2 (10.1 – 33.40)	27.0	8.0 to 50.0	
Hiltermann <sup>2</sup>	1998	Leiden	1994 – 1995	NR	3.40	0.6 to 6.8	
Hiltermann <sup>3</sup>	1998	Leiden	1994 - 1995	NR	4.80	0.2 to 6.8	

<sup>1</sup> Cough. <sup>2</sup> Shortness of breath. <sup>3</sup> Difficulty breathing upon awakening. NR= Not reported.

A final parameter, which has been associated directly with high levels of  $\text{PM}_{10}$  pollution and indirectly with the toxic effects resulting from exposure, is child absenteeism from school. Of the very few reports that have been published on this parameter, Table 8 presents two which show increases in absenteeism associated with  $\text{PM}_{10}$  pollution levels. The large disparity between the reported increases is immediately apparent. One study registered an increase of only 1% while the second reported an increase of greater than 50%. As information on exposure levels is unavailable for the Peters *et al.* study, it is impossible to determine whether this factor would explain the large difference in reports. However, the study was performed for asthmatic children under medical treatment, while the Ransom *et al.* study considered apparently healthy children. As discussed above, there tend to be significant differences in percent change for the observed variables between healthy individuals and asthmatics under medical treatment.

Table 8. **Percent change in child school absenteeism for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**

Author	Year	Location	Period	PM <sub>10</sub> levels	% change	95% CI	
				Mean (range)		LL	UL
Peters <sup>1</sup>	1997 <sup>c</sup>	Sokolov	1991 – 1992	N.R.	52.30	15.38 to 76.15	
Ransom	1992	Utah	1985 – 1990	50.0 ( NR - 365.0)	0.21	0.25 to 0.67	

<sup>1</sup>Use of medications, NR = Not reported.

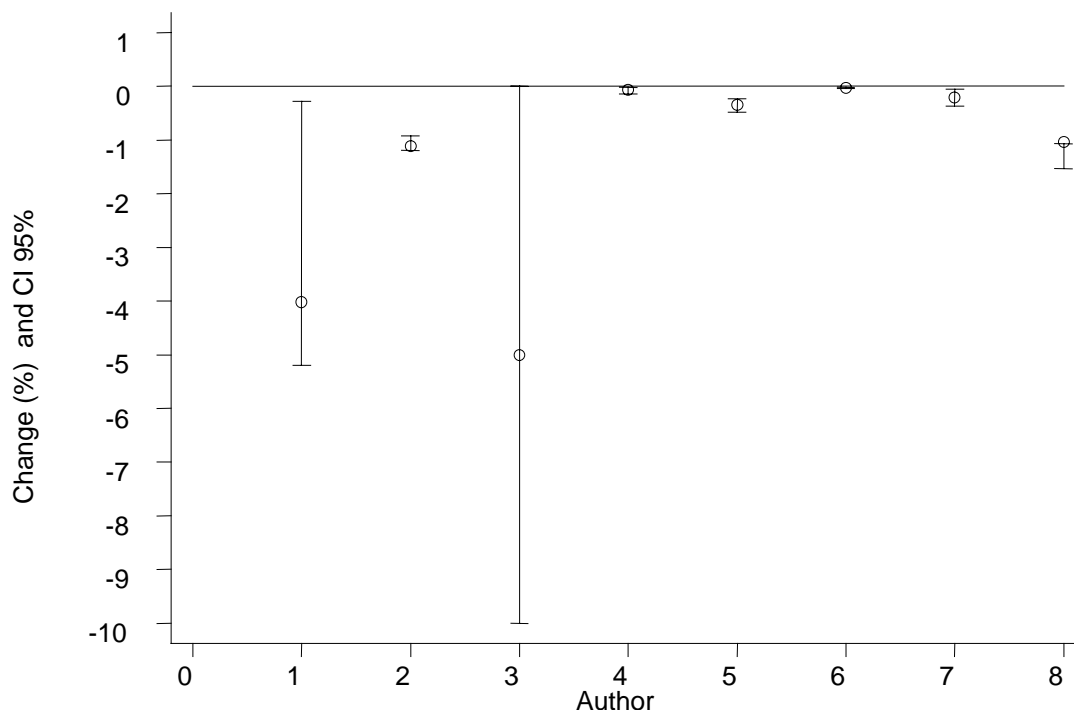
g) *Percent change in FEV-1, FVC, PEF y MMEF*

The presence of symptoms or occurrence of certain diseases is not the only parameter used to determine air pollution toxicity. It is often advisable to define some diagnostic technique that can detect toxic effects prior to the appearance of clinically recognized symptoms. Spirometric parameters represent just such a tool and have been used by associating pollution levels to forced expiratory volume at first second, (FEV-1), forced vital capacity (FVC), maximal mid-expiratory flow (MMEF) and peak expiratory flow (PEF).



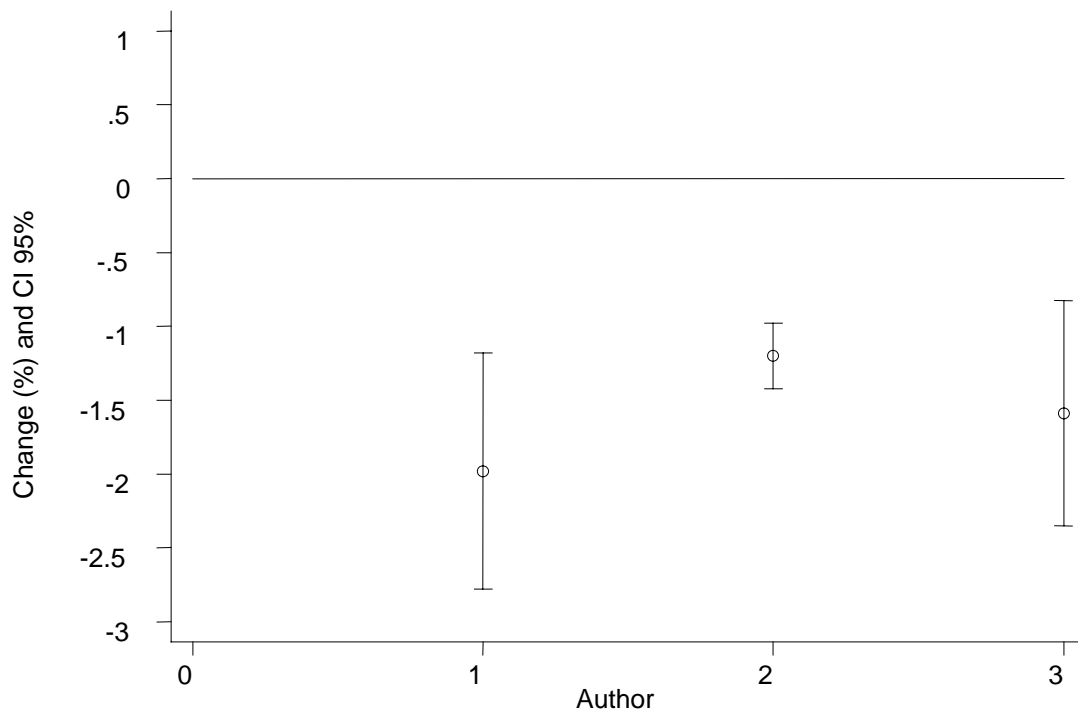
Figure 21 shows the percent change in FEV-1. In general, except for the studies by Brunekreef (4.02%), Dockery (5.00%) and Chesnut (1.12%), the reported values show percent decreases of between 0.06 and 0.98%. However, the first two values mentioned above were performed measuring FEV-0.75, which could explain the different results. Evaluation of these studies provided a very small pooled estimated decrease and a very broad confidence interval.

Figure 21. **Percent absolute change in FEV with 95% CI for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$**



Note: The numbers represent the following studies: 1. Brunekreef 1991 (Steubenville), 2. Chestnut 1991 (U.S.A.), 3. Dockery 1982 (Steubenville), 4. Hoek 1993 (Holland), 5. Hoek 1993 (Wageningen), 6. Koenig 1993 (Seattle), 7. Pope 1993 (Salt Lake City), 8. Pooled estimate.

Figure 22. Absolute percent change in FVC for each  $10 \mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$  with 95% CI

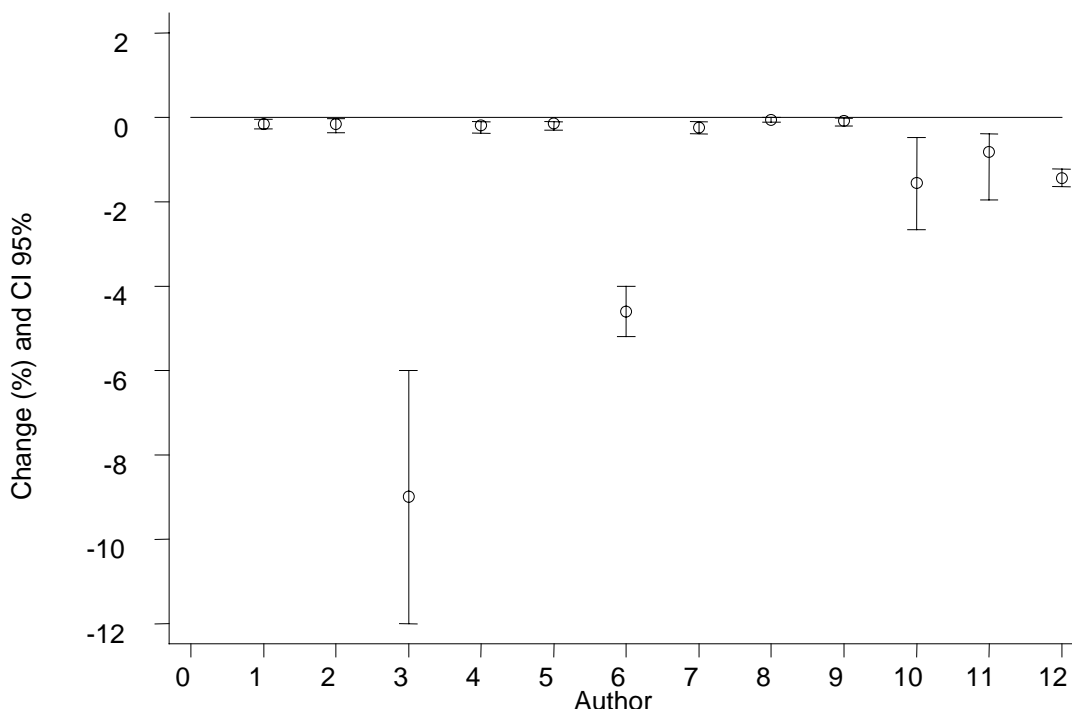


Note: The numbers represent the following studies: 1. Brunekreef 1991 (Steubenville), 2. Chestnut 1991 (U.S.A.), 3. Pooled estimated.

Fewer studies are available documenting the toxic effects of  $\text{PM}_{10}$  associated with FVC than for the previous parameter. Figure 22 shows the two major studies determining the effects of this pollutant on this spirometric diagnosis. Both studies were carried out in the U.S., and the absolute percent change in pulmonary function is very similar between the two (-1.30 and -1.58%). The pooled estimated was -1.30%, (CI 95% -1.53 to -1.07).

The association between pollutant levels and the PEF parameter has been widely documented recently and Figure 23 summarizes these studies. The results show that except for the studies by Hoek, 1994 (9.0%), Peters, 1997 (4.6%), Gold, 1999 (1.56%) and Romieu, 1996 (1.2%), the changes reported are not greater than 0.39%.

Figure 23. Absolute percent change in PEF for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$  with 95% CI



Note: The numbers represent the following studies: 1. Hoek 1993 (Holland), 2. Hoek 1993 (Wageningen), 3. Hoek 1994 (Holland), 4. Neas 1992 (Uniontown), 5. Neas 1996 (Pennsylvania), 6. Peters 1997<sup>c</sup> (Erfurt), 7. Pope 1991<sup>a</sup> (Utah), 8. Pope 1992 (Utah), 9. Roemer 1993 (Wageningen), 10. Gold 1999 (Mexico), 11. Romieu 1996 (Mexico), 12. Pooled estimate.

MMEF is the least documented of the pulmonary function diagnoses for association with pollution levels. Only one report was found to evaluate this spirometric parameter. The study, performed in Holland is summarized in Table 9.

Table 9. Percent change in MMEF<sup>1</sup> for each 10  $\mu\text{g}/\text{m}^3$  increase in  $\text{PM}_{10}$  estimated in time series studies

Author	Year	Location	Period	PM <sub>10</sub> levels	% decrease	IC 95%	
				Mean (range)		LL	UL
Hoek	1994	Holland	1987 – 1990	44.9 (14.1 - 126.1)	-8.00	-11.00	-5.0

<sup>1</sup>Maximal mid-expiratory flow.

*h) Percent change in chronic effects*

The effects of long term exposure to air pollutants on human health are extremely important since the majority of people living in urban environments are permanently exposed to low concentrations of these pollutants. Quantitative determination of such exposure is difficult given the characteristics of cities, themselves, the long term temporal-spatial pollutant distribution and varying individual patterns of activity, transit and occupations inherent in the urban environment. For all of these reasons, few studies have achieved evaluations of this type of exposure.

For studies of chronic respiratory effects associated with pollutant levels, Abbey *et al.* 1993 describes percents of change in occurrence of respiratory symptoms according to variations in PM<sub>10</sub> levels to 3.6% (CI 95% 6.6 – 1.1).

There are a few reports that find significant effects on mortality due to chronic effects; Dockery in 1993 in Ohio, found a 5.70% (95% CI, 10.44-1.7%) and Pope in 1995 in USA found 3.84% (95% CI, 2.93 – 6.75), the pooled estimated for these studies was 4.97 (95% CI 3.19 - 6.75).

*i) Percent change in restricted activity days and minor restricted activity days*

From an economical point of view the days that a worker stops his labour also called restricted activity days (RAD), or his productivity going down, denominated minor restricted activity days (MRAD), because of a sickness, represent an important factor, since this time as traduce like lost of monetary ingress. That is why it is important to quantify RAD or MRAD and the economical weight that these factors represent. In this case the Table 10 shown the percent change on RAD and MRAD.

**Table 10. Percent change for RAD and MRAD for each 10 µg/m<sup>3</sup> increase in PM<sup>10</sup>**

<b>Author</b>	<b>Locality</b>	<b>Year</b>	<b>Period</b>	<b>Parameter</b>	<b>Best estimate</b>
Ostro	USA	1990	1979-1981	DAR	7.74%
Ostro	USA	1980	1976 – 1986	DAR	9.48%
Ostro	USA	1989	1976 – 1986	DARM	4.92%

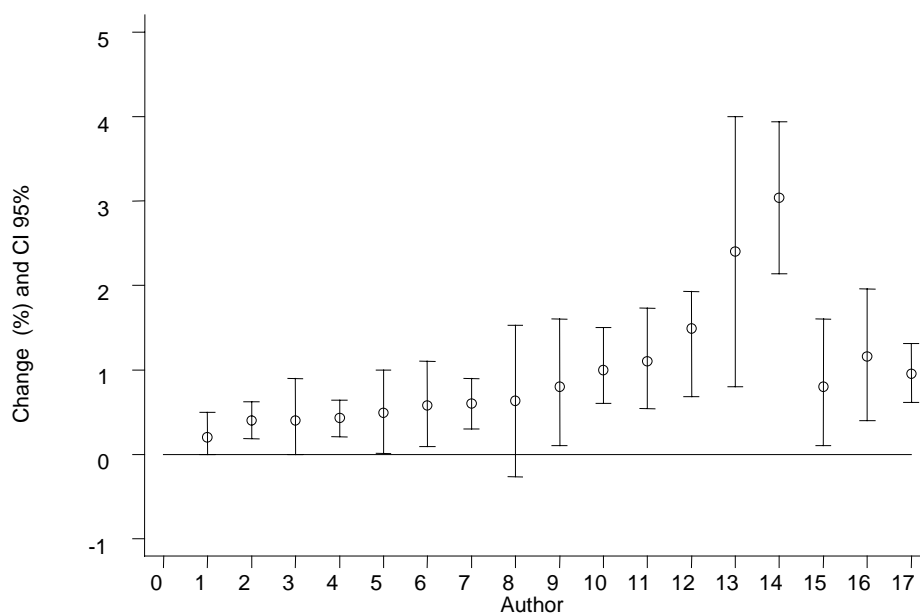
## 4.2 *Meta-analysis of health effects due to Ozone exposure*

### a) *Percent change in mortality due to ozone exposure*

The increase in mortality is one of the most significant parameters in determining the impact of a pollutant on the health of a population. However, in the case of ozone, in contrast to particulate exposure these endpoints have been debated, mainly because most of the time the pollutants are present at the same time. Trying to establish the weight of the particulate matter in the associations between ozone and total mortality we performed an evaluation considering models adjusted and not adjusted for particulate matter. Figure 25 shows studies, not adjusted by particulate matter, where an increase in total mortality (excluding accidental death) was evaluated.

Figure 24 shows studies where an increase in total mortality (excluding accidental death) was evaluated. The increases reported in these studies is low (0.2 to 1.49%), although two other studies published in the 1970's report increases as high as 2.4% and 3.04%, and pooled estimate was 0.995% per 10ppb (CI 95% 0.62 - 1.31).

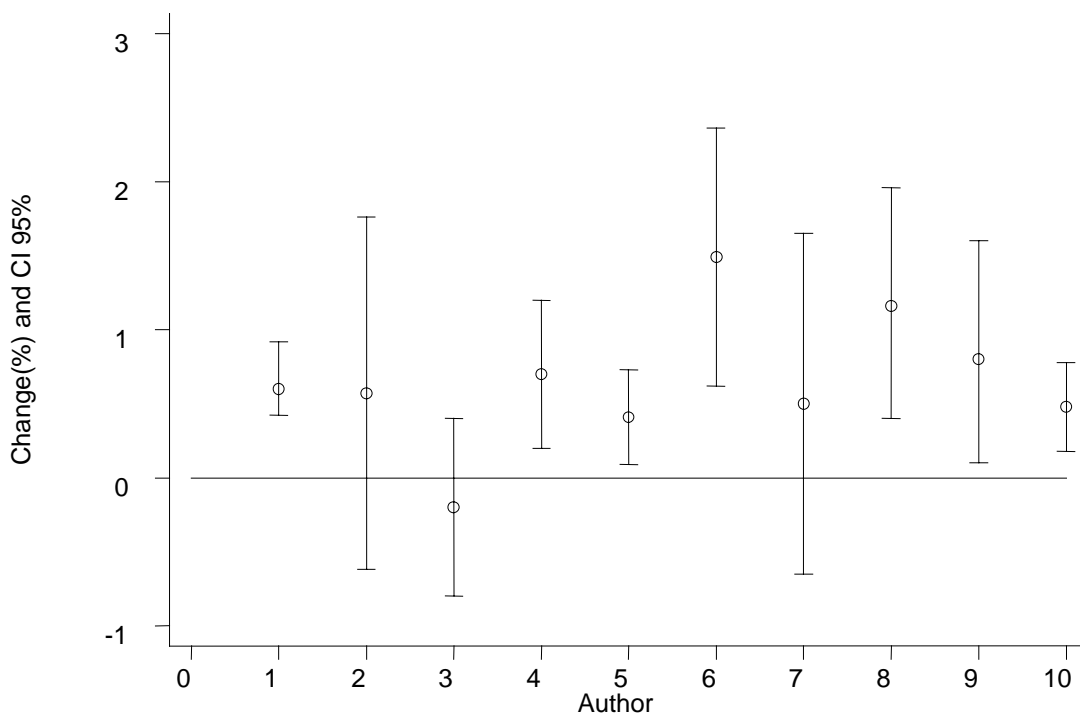
Figure 24. **Percent change in total (non-accidental) death with 95% CI for each 10 ppb increase in ozone (not adjusted for particulate matter)**



Note: The numbers correspond to the following studies: 1. Kinney 1995 (Los Angeles), 2. Loomis 1996 (Mexico), 3. Ostro 1996 (Santiago), 4. Borja 1997 (Mexico), 5. Verhoeff 1996 (Amsterdam), 6. Sunyer 1996 (Barcelona), 7. Anderson 1996 (London), 8. Borja 1998 (Mexico), 9. Hoek 1997 (Rotterdam), 10. Ito 1996 (Chicago), 11. Kelsall 1997 (Philadelphia), 12. Moolgavkar 1996a (Philadelphia), 13. Simpson 1997 (Brisbane), 14. Kinney 1991 (Los Angeles), 15. Hoek 1997 (Rotterdam), 16. Toloumi 1997 (APHEA cities), 17. Pooled estimate.

Particulate matter and ozone are often correlated spatially and over time, making it difficult to separate the effects of the individual pollutants. Thus, it could be unclear how much each pollutant may individually influence elevated mortality and morbidity rates. As a result, some cost-benefit studies have chosen one index air pollutant, rather than estimating effects for multiple air pollutants individually and then adding their effects to get a total air pollution effect. The focus on a single pollutant provides a conservative approach to estimating air pollution effects. In fact, recent analyses (e.g., Thurston and Ito, 1999) suggest that ozone and PM air pollution effects are relatively independent, since controlling for one pollutant has only modest effects on the concentration-response of the other. Thus, the use of a single index pollutant underestimates the overall public health effects and monetary valuations of air pollution changes. Recognizing that the effect of ozone on mortality independent of particulates is still on debate, we re-evaluated the effect of ozone restricting the analysis to those studies that controlled for particles in the statistical analysis

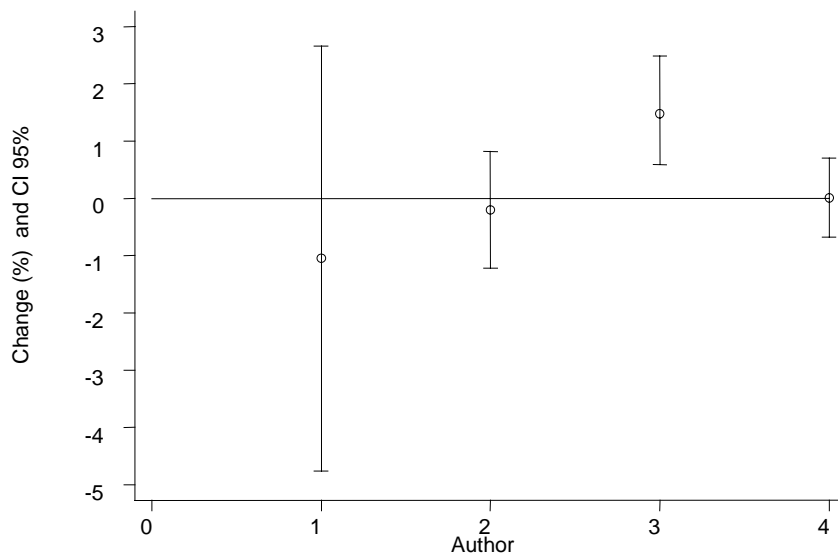
Figure 25. **Percent change in total (non-accidental) death with 95% CI for each 10 ppb increase in ozone**



Note: The numbers correspond to the following studies: 1. Anderson 1996 (London), 2. Borja 1998 (Mexico), 3. Borja 1997 (Mexico), 4. Ito 1996 (Chicago), 5. Kelsall 1997 (Philadelphia), 6. Moolgavkar 1996a (Philadelphia), 7. Verhoeff 1996 (Amsterdam), 8. Toloumi 1997 (APHEA cities), 9. Hoek 1997 (Rotterdam), 10. Pooled estimate.

Figure 25 shows the studies where an association between ozone and total mortality (non accidental) adjusting for particulate matter .In this case the increases reported in these studies are lower than the ones presented in the figure 25 (-0.2 to 1.49%). Also the pooled estimated is lower than (0.59% per 10ppb for this studies, CI 95% 0.30 - 0.86).

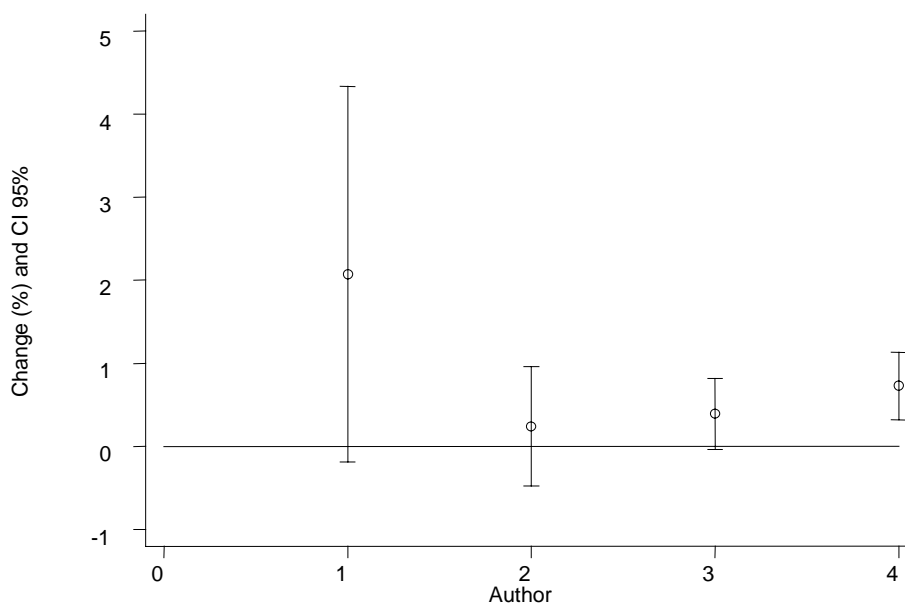
Figure 26. **Percent change in mortality due to respiratory disease with 95% CI for each 10 ppb increase in ozone**



Note: Numbers correspond to the following studies: 1. Anderson 1996 (London), 2. Borja 1997 (México), 3. Borja 1998 (México) 4. Pooled estimate.

However, determining total mortality (not accidental) can be considered a non-specific parameter. Deaths that could be related to the route of exposure, such as those due to respiratory or cardiac ailments, must also be considered. Considering respiratory ailments, only a few articles have established an association between ozone exposure and mortality (Figure 26). These studies were performed in Europe and Mexico. For these studies the pooled showed insignificant effects.

Figure 27. **Percent change in mortality due to cardiovascular disease with 95% CI for each 10 ppb increase in ozone**



Note: Numbers correspond to the following studies: 1. Anderson 1996 (London), 2. Borja 1997 (México), 3. Borja 1998 (México) 4. Pooled estimate.

Mortality due to cardiac ailments the highest percent change of 1.76% is found in the Borja *et al.* 1998 study, determined an elevated increase of risk. Here the pooled estimated shown in Figure 27 was 0.73 (CI 95% 0.31 - 1.13).

Individuals older than 65 years of age should be studied independently from the rest of the population since this group could be at increased risk because of reduced systemic defences against pollution's toxic effects. Table 11 summarises studies that evaluated mortality for this population. Both studies were non-significant.

Table 11. **Effects of O<sub>3</sub> on mortality in individuals 65 years or older for each 10 ppb increase in ozone**

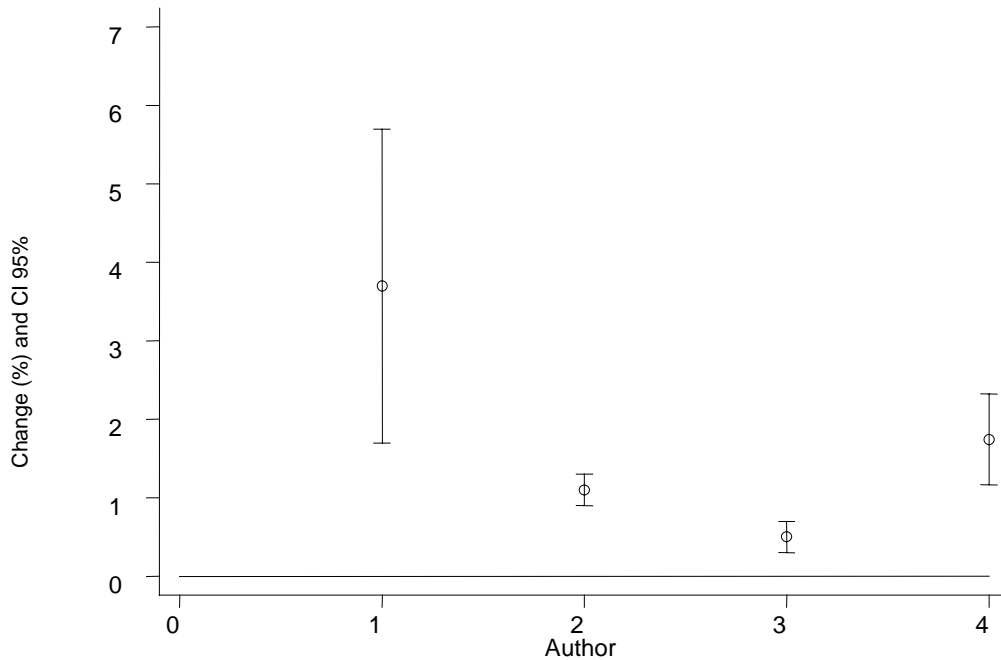
Author	Year	Locality	Period	Mean	IC 95%
Borja	México	1998	1993 - 1995	0.07	-2.15 – 2.29
Borja	México	1997	1990 - 1992	-0.10	-0.56 – 0.36



b) *Percent change in total hospitalisations*

Total hospital admissions (non-accidental) represents another parameter for which measuring pollution impact. Studies evaluating change in total admissions are summarised in Figure 28 of this section. The studies were performed in the United States and Canada, with the largest percent change registered in Buffalo, New York (3.70%). The pooled estimated change was 1.74% (CI 95% 1.16 - 2.32).

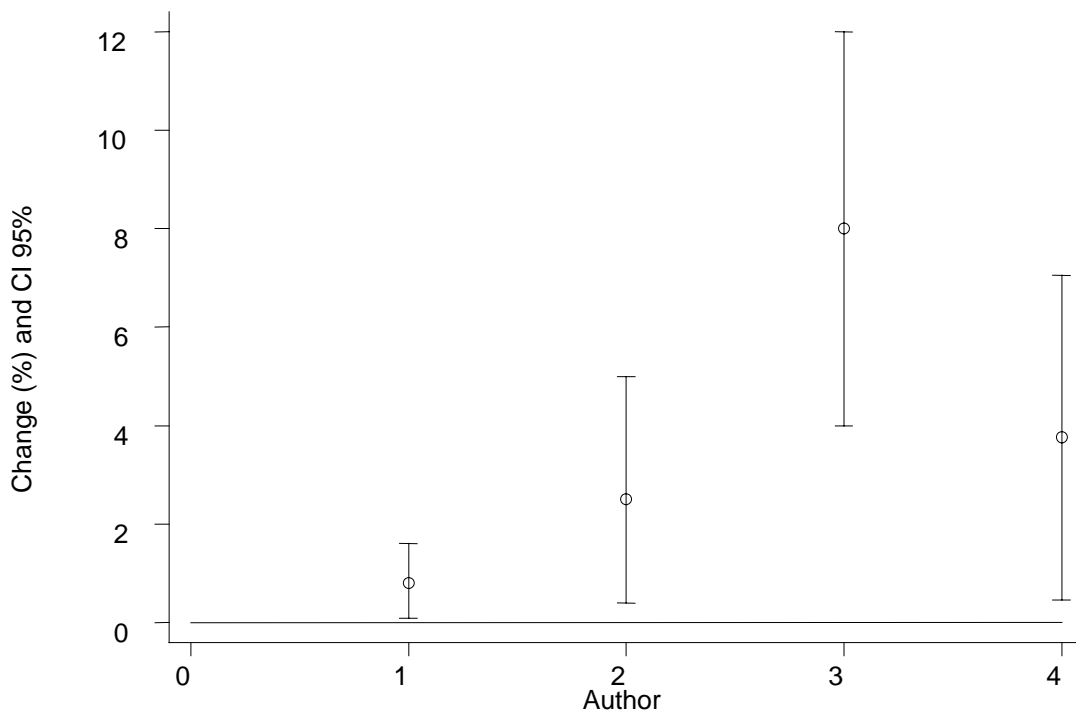
Figure 28. **Percent change in total hospitalisations with 95% CI for each 10ppb increase in ozone**



Note: The numbers represent the following studies: 1. Thurston 1992 (Buffalo), 2. Thurston 1992 (New York), 3. Thurston 1994 (Ontario), 4. Pooled estimate.

Total hospitalisation is a fairly non-specific parameter for determining toxic effects due to ozone contamination. It is therefore necessary to deal with specific causes of hospitalisation, again, concentrating on those causes which could be related to exposure route (hospitalisations for respiratory and cardiac ailments).

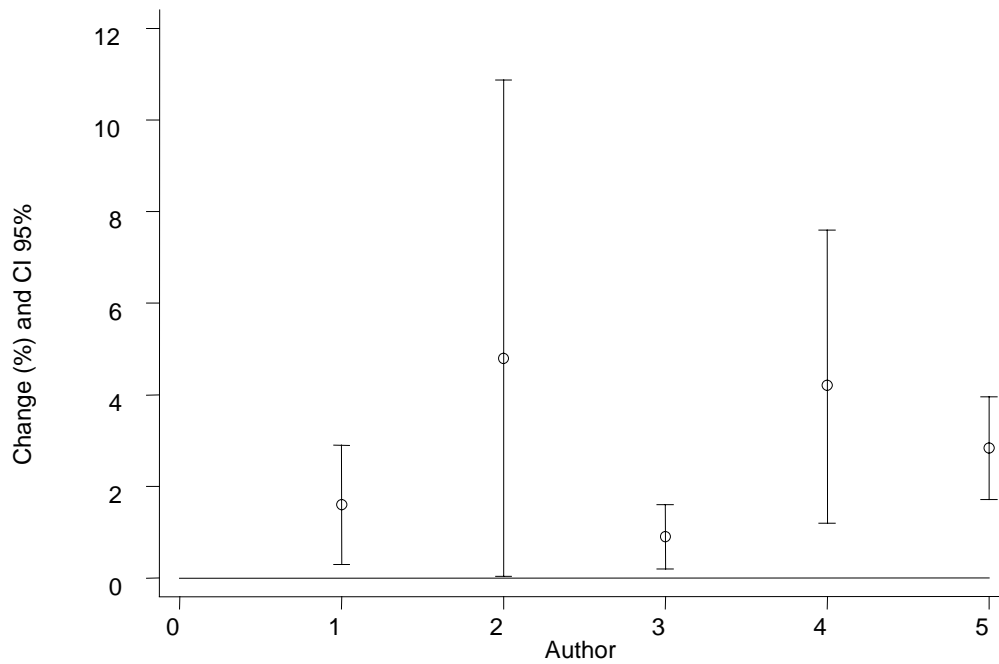
Figure 29. **Percent change in hospitalisations due to respiratory diseases with 95% CI for each 10 ppb increase in ozone**



Note: The numbers represent the following studies 1. Ponce de León 1996 (London), 2. Schouten 1996 (Rotterdam), 3. Linn 2000 (Los Angeles) 4. Pooled estimate.

Figure 29 presents the results from two studies where a significant increase in hospitalisation was reported for respiratory diseases. The studies were carried out in developed American and European countries. Percent increases between 0.8 and 8 % were reported. The weighted average increase in specific hospitalisations for respiratory diseases 3.76% (CI 95% 0.45 - 7.05).

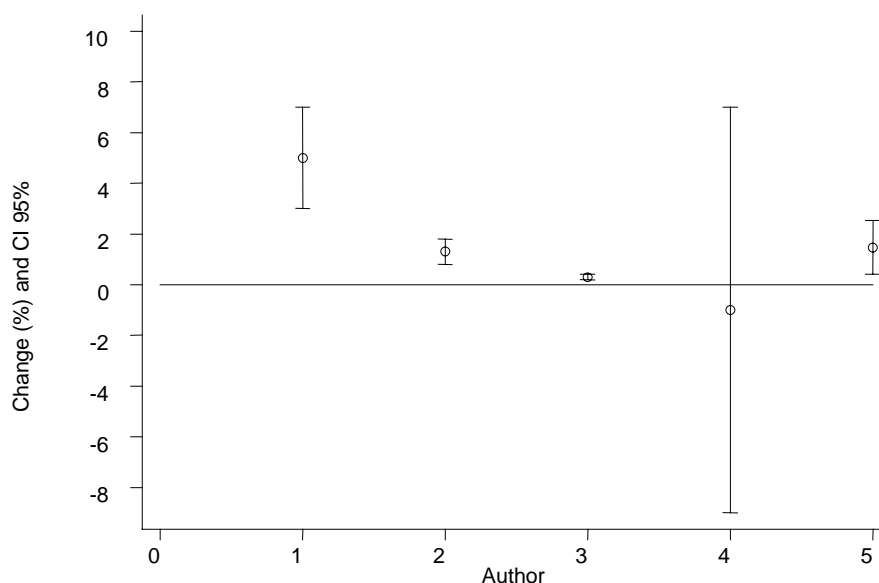
Figure 30. **Percent change in hospitalisations for individuals older than 65 years due to respiratory disease CI-95% for each 10 ppb increase in ozone**



Note: Asthma is one of the respiratory ailments for which an increase in hospitalisations has been observed.

Figure 30 shows the results of hospital admissions for respiratory diseases for individuals older than 65 years. The highest pooled estimated was 2.83 (CI 95% 1.71 – 3.95).

Figure 31. **Percent change in hospitalisations due to asthma with 95% CI for each 10 ppb increase in ozone**



Note: The numbers represent the following studies: 1. Thurston 1992 (Buffalo), 2. Thurston 1992 (NY), 3. Thurston 1994 (Ontario), 4. Linn 2000 (Los Angeles), 5. Pooled estimate.

Figure 31 presents the results of some of the most important studies where an increase in hospitalisation has been reported for this illness. Again, the city of Buffalo, New York, registers the highest increase of 5.0%. The pooled estimated increase was 1.47% (CI 95% 0.41 - 2.53).

Hospitalisations for chronic obstructive pulmonary disease (COPD) and pneumonia are two other factors that increase with exposure to ozone. Only two reports were found in Detroit, Illinois, and Minneapolis, Minnesota, where both diseases were studied. The percent change for COPD was similar to the pneumonia, between 4.2 and 5.5% increase in COPD, and 5.2 and 5.7% increase in pneumonia.

Table 12. **Percent change in hospitalisations for COPD for each 10 ppb increase in ozone**

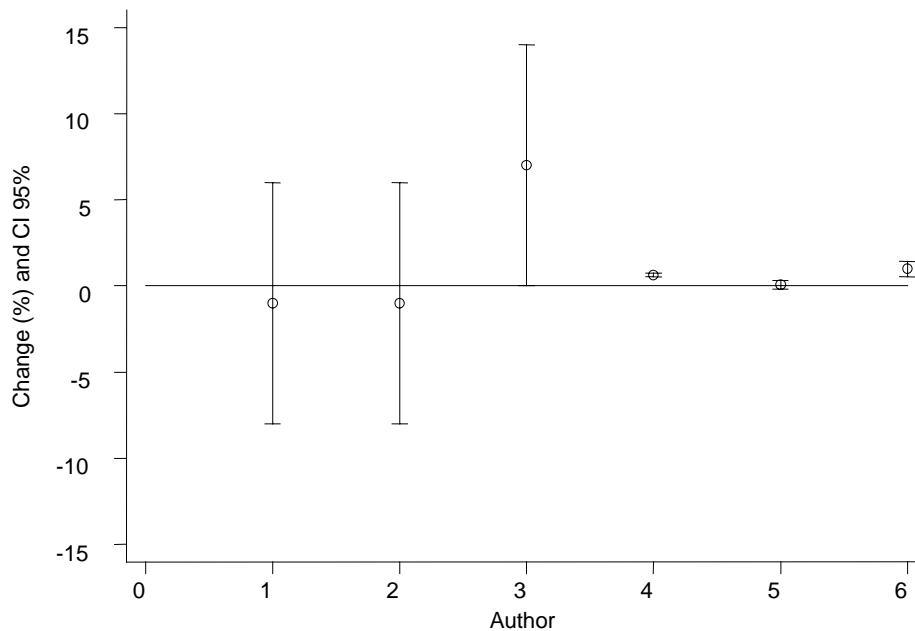
Author	Locality	Year	Period	Mean	IC 95%
Schwartz	Detroit	1994	1986 - 1989	5.5	-3.4 - 7.5
Moolgavcar	Minneapolis	1997	1986 - 1991	4.2	-1 - 9.4

Table 13. **Percent change in hospitalisations for individuals older than 65 years due to pneumonia for each 10 ppb increase in ozone**

Author	Locality	Year	Period	Mean	IC 95%
Schwartz	Detroit	1994	1986 - 1989	5.2	2.6 – 8.0
Moolgavcar	Minneapolis	1997	1986 - 1991	4.7	-5.2 – 9.0

Finally, changes in hospitalisations due to cardiovascular disease is an important end point, some of the most important studies are presented in the Figure 32 where occlusive stroke has the biggest change with a positive percent (7.00%). The pooled estimate was 0.98% (CI 0.53 - 1.43).

Figure 32. **Percent change in hospitalisations due to cardiovascular disease with 95% CI for each 10 ppb increase in ozone**



Note: All the plots belong to Linn 2000 (Los Angeles) study with the following diagnostics: 1. Congestive Heart failure, 2. Cardiac arrhythmia, 3. Occlusive stroke, 4. Total cardiovascular, 5. Cerebrovascular, 6. Pooled estimate.

c) *Percent change in hospital emergency room admissions*

Increase in hospital emergency room admissions for respiratory ailments and asthma represents another useful parameter for studying the toxic effects of ozone exposure. Results of studies evaluating this parameter are summarised in Tables 14 and 15. It is immediately obvious that the values differ widely between populations. For respiratory diseases, the pooled estimate for general population increase was 3.172% (CI 95% 1.672 - 4.671) and for asthma from 3.5 to 15.0%. A study in Mexico City reported an increase of 20% in emergency room admission for diagnosed tracheitis (Table 16).

Table 14. **Percent change in emergency room admissions for respiratory diseases for each 10 ppb increase in ozone**

<b>Author</b>	<b>Locality</b>	<b>Year</b>	<b>Period</b>	<b>Mean</b>	<b>IC 95%</b>
Delfino	Montreal	1997	1992 - 1993	7.22	2.9 – 11.5
Tellez-Rojo	México	1997	1993	1.98	1.40- 2.58
Burnett	Ontario	1998	1994	0.42	-0.38- 1.22

Table 15. **Percent change in emergency room admissions for asthma for each 10 ppb increase in ozone**

<b>Author</b>	<b>Locality</b>	<b>Year</b>	<b>Period</b>	<b>Mean</b>	<b>IC 95%</b>
Damakosh	México	2000	1993 - 1994	15	0 – 40.0
Stieb	New Brunswick	1996	1984 - 1992	3.5	1.7 – 5.3

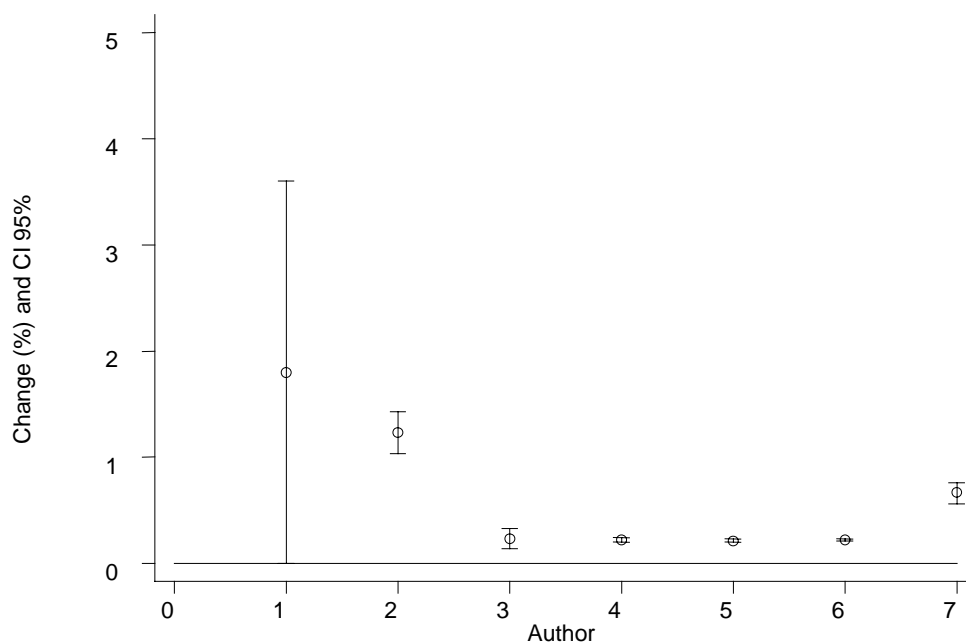
Table 16. **Percent change in emergency room admissions for tracheitis for each 10 ppb increase in ozone**

<b>Author</b>	<b>Locality</b>	<b>Year</b>	<b>Period</b>	<b>Mean</b>	<b>CI 95%</b>
Damakosh	Mexico City	2000	1993 – 1994	20.0	0.00 - 40.0

d) *Percent change in different respiratory symptoms in asthmatic children*

Asthmatic children are can be more sensitive to the effects of contamination and thus must be considered as another group worthy of special attention. Figure 33 of this section presents the association between increased asthmatic attacks and average ozone concentrations. The results show an increase in the presence of respiratory diseases of 0.66% (CI 95% 0.55 - 0.76) was also reported for this group. These figures reaffirm what we already know about this population which suffers the effects of pollution to a much greater extent than others.

Figure 33. **Percent change with 95% CI in different respiratory symptoms in asthmatic children for each 10 ppb increase in ozone**



Note: The numbers represent the following studies: 1. Hiltermann 1998 (Leiden) (Difficulty breathes), 2. Gielen 1997 (Amsterdam) (upper respiratory symptoms), 3. Peters 1999 (California) (Cough), 4. Romieu 1996 (Mexico) (upper respiratory symptoms), 5. Romieu 1996 (Mexico) (lower respiratory symptoms), 6. Romieu 1996 (Mexico) (Difficulty breathes), 7 Pooled estimate.

Table 17 shows other studies with increase of 2.45% to 5% in asthmatic attacks reported by Ostro *et al.* 1995<sup>a</sup> and Dockery 1989 followed by a 1.80% increase in asthmatic attacks (where a bronchial-dilator was used) in the Hiltermann report and the presence of lower respiratory tract symptoms 0.23%.

Table 17. **Percent change in different respiratory symptoms in asthmatics for each 10 ppb increase in ozone for each 10 ppb increase in ozone**

Author	Locality	Year	Period	Mean	95 % CI
Peters <sup>1</sup>	California	1999	1986 - 1990	0.23	0.14 - 0.33
Ostro <sup>2</sup>	Los Angeles	1995	1992	5.00	1.75 - 9.00
Dockery <sup>2</sup>	U.S.A.	1989	1980 - 1981	2.45	0.0 - 5.9
Hilterman <sup>3</sup>	Leiden	1998	1995	1.80	0.20 - 3.60

<sup>1</sup> Lower respiratory tract symptoms, <sup>2</sup> Asthmatic attacks, <sup>3</sup> Asthmatic attacks and use of bronchial-dilator.

e) *Percent change in different respiratory symptoms for the general population*

Besides increased hospitalisations and emergency admissions, an association has been observed between ozone exposure and the increase of certain diseases. Although few publications document this association, existing data report an increase in lower (Table 18) and upper (Table 19) respiratory tract diseases and wheezing episodes in children (Table 20). A possible reason for the scant literature reporting these types of associations is that more emphasis has been placed on studying the special, high-risk populations.

Table 18. **Percent change in lower respiratory tract symptoms for each 10 ppb increase in ozone**

<b>Author</b>	<b>Locality</b>	<b>Year</b>	<b>Period</b>	<b>Mean</b>	<b>95 % CI</b>
Ostro	California	1993	1978 - 1979	2.2	1.1 – 3.4
Olaiz	México	2000	1996 - 1997	1.1	-0.3 – 2.4

Table 19. **Percent change in wheezing for each 10 ppb increase in ozone**

<b>Author</b>	<b>Locality</b>	<b>Year</b>	<b>Period</b>	<b>Mean</b>	<b>95 % CI</b>
Buchdahl	Londres	1996	1992 - 1993	1.32	0.47 – 2.4
Hiltermann	Leiden	1998	1995	4.4	1.0 – 8.8

Table 20. **Percent change in upper respiratory tract symptoms for each 10 ppb increase in ozone**

<b>Author</b>	<b>Locality</b>	<b>Year</b>	<b>Period</b>	<b>Mean</b>	<b>95 % CI</b>
Olaíz	Mexico	2000	1996 – 1997	1.5	1.1 – 2.2

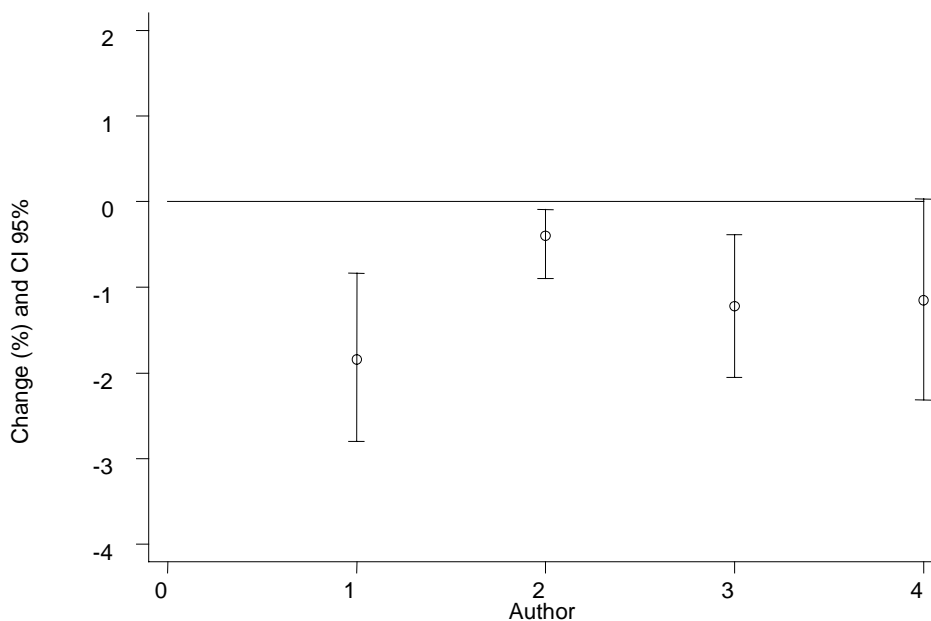
f) *Percent change in PEF, FEV-1 and FVC*

A commonly employed strategy to establish the toxic effects of ozone at low concentration is the measurement of spirometric parameters such as peak expiratory flow (PEF), forced expiratory volume per second (FEV-1) and the forced vital capacity (FVC).



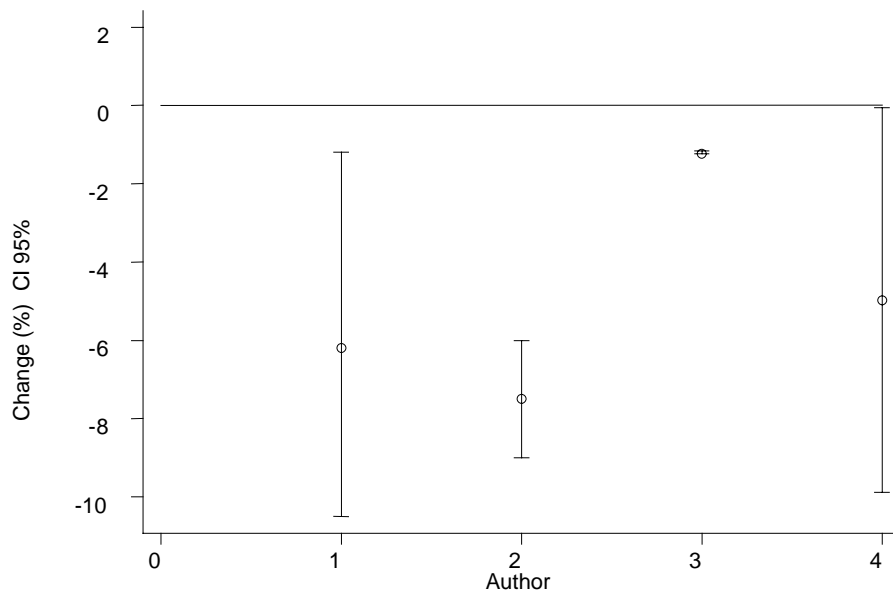
Figures 34, 35 and 36 in this section summarise the reports relating changes in PEF, FEV-1 and FVC. Among the three parameters, the PEF was noticeably less with an range of values between -0.4 and -1.84%. The decreases found for FEV-1 were between -6.2 and -7.5%. The range of values for FVC between -6.0 and -7.2.

Figure 34. **Percent change with 95% CI of PEF with each 10 ppb increase in ozone**



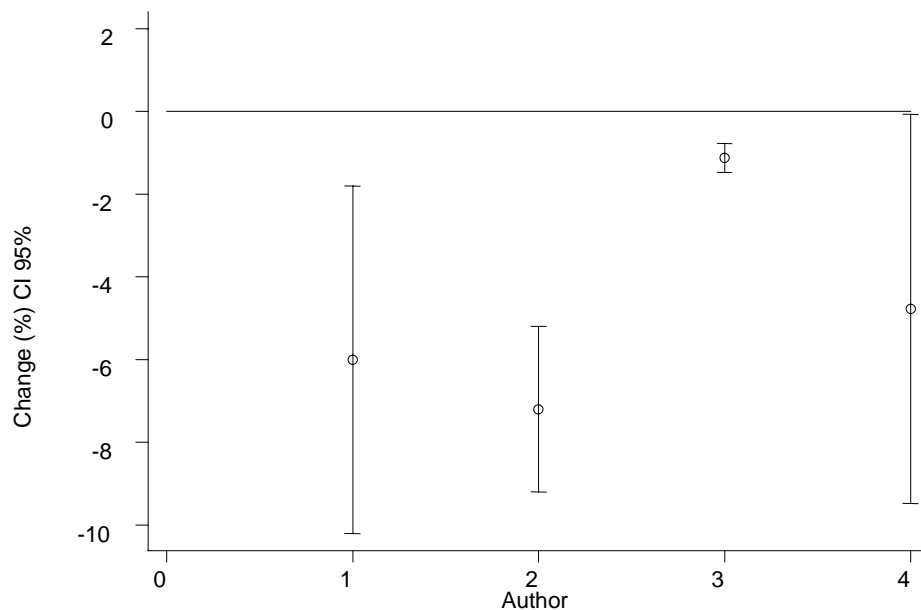
Note: The numbers represent the following studies: 1. Gold 1999 (Mexico), 2. Romieu 1996 (Mexico), 3. Castillejos 1992 (México), 4. Pooled estimated.

Figure 35. **Percent change with 95% CI for FEV-1 for each 10 ppb increase in ozone**



Note: The numbers represent the following studies: 1. Brunnekreef 1991 (Kingston), 2. Brunnekreef 1991 (New Jersey), 3. Castillejos 1992(México), 4. Pooled estimate.

Figure 36. **Percent change with 95% CI for FVC for each 10 ppb increase in ozone**



Note: The numbers represent the following studies: 1. Brunnekreef 1991 (Kingston), 2. Brunnekreef 1991 (New Jersey), 3. Castillejos 1992 (México), 4. Pooled estimate.

g) *Percent change in restricted activity days and minor restricted activity days*

Table 21 present the percent change on RAD and MRAD where just for the first one, there are significative results.

**Table 21. Percent change for RAD and MRAD for each 10 ppb increase in ozone**

<b>Author</b>	<b>Locality</b>	<b>Year</b>	<b>Period</b>	<b>Parameter</b>	<b>Best estimate</b>
Ostro	USA	1980	1976 – 1986	RAD	18.5%
Ostro	USA	1989	1976 – 1986	MRAD	0

## 5. Summary Tables

Pooled estimated of the Health Effects of PM <sub>10</sub> and O <sub>3</sub>	Ozone		PM <sub>10</sub>	
	Mean <sup>1</sup>	IC 95%	Mean <sup>2</sup>	IC 95%
<i>1.1 Acute Mortality (adjusted by particles)</i>				
Total mortality	0.59 (9)	0.30 - 0.86	1.01 (39)	0.83 - 1.18
Respiratory mortality	0.01 (3)	-0.68 - 0.70	1.82(21)	1.37 - 2.22
Cardiovascular mortality	0.73 (3)	0.32 - 1.13	1.32(20)	1.10 - 1.55
> 65 mortality	0.07 (1)	-2.15 - 2.29	1.18 (11)	0.66 - 1.57
Infant mortality	-	-	3.52 (1)	0.72 - 6.31
<i>1.2 Hospital admissions</i>				
Total	1.74 (3)	1.16 - 2.32		
Respiratory hospital admissions	3.76 (3)	0.45 - 7.05	1.39 (11)	1.18 - 1.60
Respiratory hospital admissions (>65)	2.83 (4)	1.71 - 3.95	1.49 (6)	1.20 - 1.78
Asthma hospital admissions	1.47 (4)	0.41 - 2.53	3.02 (11)	2.05 - 4.00
ECOP hospital admissions	5.50 (1)	-3.40 - 7.50	2.34 (11)	1.80 - 2.89
Neumonia hospital admissions	5.20 (1)	2.60 - 8.00	1.40 (4)	1.05 - 1.75
Cardio & Cerebro- vascular hospital admissions	0.98 (5)	0.53 - 1.43	0.60 (12)	0.42 - 0.79
Cardio & Cerebro vascular hospital admissions (>65)	-	-	1.22 (4)	0.94 - 1.50
<i>1.3. Emergency room visits (ERVs)</i>				
Total				
Respiratory causes	3.172 (3)	1.67 - 4.67	3.11 (5)	2.35 - 3.88
Asthma causes	3.50 (1)	1.70 - 5.30	4.50 (1)	2.16 - 7.00
Tracheitis	12.5 (1)	0.0 - 29.16	-	-

<sup>1</sup> Percent change per 10 ppb. <sup>2</sup> Percent change per 10 µg/m<sup>3</sup>

The number in parenthesis are the studies included in the calculation of the pooled estimated.

	Ozone		PM <sub>10</sub>	
	Mean <sup>1</sup>	IC 95%	Mean <sup>2</sup>	IC 95%
<i>1.4 Effects in asthmatics</i>				
Asthma attacks	2.45 (1)	0.00 – 5.90	7.87 (7)	4.48 – 11.27
Asthma attacks & Bronchodilator usage	1.80 (1)	0.20 - 3.60	10.22 (6)	7.30 – 13.14
Cough without phlegm (Children)	-		4.54 (5)	2.65 – 6.44
Cough with phlegm (Children)			3.32 (2)	2.01 – 4.64
Some respiratory symptoms (Children)	0.66 (6)	0.55 - 0.76		
Lower respiratory symptoms	0.23 (1)	0.14 - 0.33		
<i>1.5 Respiratory symptoms in the General Population</i>				
Respiratory symptoms			7.72 (3)	0.61 – 14.84
Upper respiratory symptoms	1.50 (1)	1.10 – 2.20	4.39 (3)	3.56 – 5.12
Lower respiratory symptoms	2.20 (1)	1.10 – 3.40	6.85 (5)	5.16 – 8.54
Bronchitis			11.00 (5)	8.96 – 13.58
Wheeze	1.32 (1)	0.47 – 2.40		
<i>1.6 Lung functions indices</i>				
PEF	-1.15 (3)	-2.32 - 0.02	-0.39 (11)	-0.48 to – 0.31
FEV	-4.97 (3)	-9.89 to –0.06	-1.30 (7)	-1.53 to –1.07
FVC	-4.77 (3)	-9.47 to –0.07	-1.58 (2)	-2.35 to –0.82
MMEF			-8.00 (1)	-5.00 to -11.00
<i>1.7 Restricted activity days and minor restricted activity days</i>				
RAD	18.50 (1)		7.74 (1)	
MRAD			4.92 (1)	
<i>2. Evaluation of Chronic Effects by Endpoint and Pollutant</i>				
2.1. Chronic Total Mortality			5.700 (1)	1.77 – 10.4
2.2 Chronic morbidity (respiratory symptoms)			3.6 00 (1)	1.10 – 6.60

Note: The number in parenthesis are the studies included in the calculation of the pooled estimated

<sup>1</sup> Percent change per 10 ppb. <sup>2</sup> Percent change per 10 µg/m<sup>3</sup>.

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# THE HEALTH BENEFITS OF CONTROLLING CARBON EMISSIONS IN CHINA<sup>75</sup>

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## 1. Introduction

Air pollution from rapid industrialization and the use of energy has been recognized to be a cause of serious health problems in urban China. For example, the World Bank (1997) estimated that air pollution caused 178,000 premature deaths in urban China in 1995 and valued health damages at nearly 5% of GDP. The same study estimated that hospital admissions due to pollution-related respiratory illness were 346,000 higher than if China had met its own air pollution standards, there were 6.8 million additional emergency room visits, and 4.5 million additional person-years were lost because of illnesses associated with pollution levels that exceeded standards. Much of this damage has been attributed to emissions of particulates and sulfur dioxide. Furthermore, this problem is expected to grow in the near future as rapid growth outpaces efforts to reduce emissions.

While particulate and sulfur dioxide emissions from burning fossil fuel contributes to local pollution, the use of fossil fuels also produces carbon dioxide, a greenhouse gas thought to be a major contributor to global climate change. The issue of climate change has engaged policy makers for some time now and is the focus of much current research.

As part of this research, in a previous paper, we examined the effects of limiting CO<sub>2</sub> emissions in China through the use of a carbon tax. In this paper we make a first attempt at estimating the local health benefits of such policies. Unlike many other efforts aimed at estimating health effects, which focus on specific technological policies to reduce pollution, here we examine broad based *economic* policies within a framework which includes all sectors of the economy. We present a preliminary effort, utilizing a number of simplifications, to illustrate the procedure. We plan to use more sophisticated air quality modelling techniques in future work.

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Our simple estimates, nevertheless, are instructive. We find that a policy which reduces carbon emissions by 5% every year from our base case will also reduce premature deaths by some 3.5 to 4.5%. If we apply commonly used valuation methods, the health damage caused by air pollution in the first year is about 5% of GDP. A policy to modestly reduce carbon emissions would therefore reduce local health losses by some 0.2% of GDP annually.

## **2. The economy-energy-health model**

Our economic modeling framework is described in Garbaccio, Ho, and Jorgenson (1999). We summarize only key features of the model here. Instead, we describe in some detail the health aspects of our model. Our approach is to first estimate the reduction in emissions of local pollutants due to policies to reduce CO<sub>2</sub> emissions. These changes in emissions are translated into changes in concentrations of various pollutants in urban areas. Dose-response functions are then used to calculate the effect of reductions in concentrations of pollutants on health outcomes. These include reduced premature mortality, fewer cases of chronic bronchitis, and other health effects. Finally, we utilize commonly used valuation methods to translate the reduced damages to health into *yuan* values which may be compared to the other costs and benefits of such policies.

### **2.1 The economic model**

Our model is a standard multi-sector Solow growth (dynamic recursive) model that is modified to recognize the two-tier plan-market nature of the Chinese economy. The equations of the model are summarized in Appendix A. As listed in Table 1, there are 29 sectors, including four energy sectors. Output is produced using constant returns to scale technology. Enterprises are given plan output quotas and the government fixes prices for part of their output. They also receive some plan inputs at subsidized prices. Marginal decisions, however, are made using the usual “price equals marginal cost” condition. Domestic output competes with imports, which are regarded as imperfect substitutes.

The household sector maximizes a utility function that has all 29 commodities as arguments. Income is derived from labor and capital and supplemented by transfers. As in the original Solow model, the private savings rate is set exogenously. Total national savings is made up of household savings and enterprise retained earnings. These savings, plus allocations from the central plan, finance national investment (and the exogenous government deficit and current account). This investment increases the stocks of both market and plan capital.

Labor is supplied inelastically by households and is mobile across sectors. The capital stock is partly owned by households and partly by the government. The plan part of the stock is immobile in any given period, while the market part responds to relative returns. Over time, plan capital is depreciated and the total stock becomes mobile across sectors.

The government imposes taxes on enterprise income, sales, and imports, and also derives revenue from a number of miscellaneous fees. On the expenditure side, it buys commodities, makes transfers to households, pays for plan investment, makes interest payments on the public debt, and provides various subsidies. The government deficit is set exogenously and projected for the duration of the simulation period. This exogenous target is met by making government spending on goods endogenous.

Finally, the rest-of-the-world supplies imports and demands exports. World relative prices are set to the data in the last year of the sample period. The current account balance is set exogenously in this one-country model. An endogenous terms of trade exchange rate clears this equation.

The level of technology is projected exogenously, i.e. we make a guess of how input requirements per unit output fall over time, including energy requirements. For the later, this is sometimes called the AEEI (autonomous energy efficiency improvement). In the model, there are separate sectors for coal mining, crude petroleum, petroleum refining, and electric power. Non-fossil fuels, including hydropower and nuclear power, are included as part of the electric power sector.

Table 1. Sectoral characteristics for China, 1992

Sector	Gross Output (bil. yuan)	Energy Use (mil. tn. coal equivalent)	Emission Height Class
1 Agriculture	909	50	low
2 Coal Mining	76	44	medium
3 Crude Petroleum	69	22	medium
4 Metal Ore Mining	24	6	medium
5 Other Non-metallic Ore Mining	66	13	medium
6 Food Manufacturing	408	36	medium
7 Textiles	380	33	medium
8 Apparel & Leather Products	149	5	medium
9 Lumber & Furniture Manufacturing	50	20	medium
10 Paper, Cultural, & Educational Articles	176	19	medium
11 Electric Power	115	49	high
12 Petroleum Refining	108	32	medium
13 Chemicals	473	138	medium
14 Building Material	254	109	medium
15 Primary Metals	321	119	medium
16 Metal Products	141	23	medium
17 Machinery	390	34	medium
18 Transport Equipment	163	5	medium
19 Electric Machinery & Instruments	155	9	medium
20 Electronic & Communication Equipment	107	2	medium
21 Instruments and Meters	24	1	medium
22 Other Industry	75	7	medium
23 Construction	520	14	low
24 Transportation & Communications	267	51	low
25 Commerce	635	14	low
26 Public Utilities	205	17	low
27 Culture, Education, Health, & Research	227	19	low
28 Finance & Insurance	171	1	low
29 Public Administration	191	7	low
Households			low
Government			-
<b>Totals</b>	<b>6,846</b>	<b>900</b>	

Source: Development Research Center Social Accounting Matrix for 1992; State Statistical Bureau; and author's estimates.

A carbon tax is a tax on fossil fuels at a rate based on their carbon content. This tax is applied to the output of three industries – coal mining, crude petroleum, and petroleum refining. It is applied to imports while exports are excluded. In the base case this tax is zero. In the policy simulations the carbon tax rate is set to achieve a desired reduction in carbon emissions. Since the application of this tax will raise revenues above those in the base case, to maintain comparability, we keep government spending and revenues the same by reducing other existing taxes.

## 2.2 *The environment-health aspects*

Emissions of local pollutants comes from two distinct sources, the first is due to the burning of fossil fuels (combustion emissions), the other from non-combustion processes (process emissions). A great deal of dust is produced in industries like cement production and building construction that is not related to the amount of fuel used. In this paper we concentrate on two pollutants, particulate matter less than 10 microns (PM-10) and sulfur dioxide (SO<sub>2</sub>). The analysis of the health effects of other pollutants, such as nitrogen oxides and lead, are left for future work. PM-10 and SO<sub>2</sub> both have their origins in combustion and non-combustion sources. Our specification of emissions, concentrations, and dose-response follows Lvovsky and Hughes (1997).<sup>76</sup>

Total emissions from industry  $j$  is the sum of process emissions and combustion emissions from burning coal, oil, and gas. Let  $EM_{jxt}$  denote the emissions of pollutant  $x$  from industry  $j$  in period  $t$ . Then we have:

$$(1) \quad EM_{jxt} = \sigma_{jx} QI_{jt} + \sum_f (\psi_{jxf} AF_{jft}) \quad ,$$

where  $x = \text{PM-10, SO}_2$  ,  $f = \text{coal, oil, gas}$  ,  $j = 1, 2, \dots, 29, \text{H, G}$  .

$\sigma_{jx}$  is process emissions of pollutant  $x$  from a unit of sector  $j$  output and  $\psi_{jxf}$  is the emissions from burning one unit of fuel  $f$  in sector  $j$ .  $QI_{jt}$  is the quantity of output  $j$  and  $AF_{jft}$  is the quantity of fuel  $f$  (in tons of oil equivalent (toe)) consumed by sector  $j$  in period  $t$ . The model generates intermediate inputs, denoted  $A_{ijt}$ , which are measured in constant *yuan*. For the cases where  $i$  is one of the fuels, these  $A_{ijt}$  's are translated to  $AF_{jft}$  which are in tons of oil equivalent of coal, oil, and natural gas. The  $j$  index runs over the 29 production sectors and the non-production sectors (household and government). For the two non-production sectors there are zero process emissions ( $\sigma_{jx} = 0$ ).

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<sup>76</sup> Lvovsky and Hughes (1997) discuss the choice of PM-10 rather than total suspended particulates (TSP). The data currently collected by the Chinese authorities are mostly in TSP. Health damage, however, is believed to be mainly due to finer particles. Lvovsky and Hughes make an estimate of the share of PM-10 in TSP and kept that constant. Improved data would obviously refine this and other analyses.

The amount of emissions per *yuan* of output, or emissions per toe of fuel used, depends on the technology employed and will change as new investments are made. A proper study should take into account the costs of these new technologies and how much they reduce emissions and energy use.<sup>77</sup> Estimates of these factors have not yet been assembled for many industries in China and we use a simple mechanism to represent such changes. Lvovsky and Hughes (1997) make an estimate of the emission levels of “new” technology and write the actual emission coefficients as a weighted sum of the coefficients from the existing and new technologies. Using superscripts “*O*” and “*N*” to denote the old and new coefficients we have:

$$(2) \quad \psi_{jxt} = k_t \psi_{jxt}^O + (1 - k_t) \psi_{jxt}^N \quad ,$$

where the weight,  $k_t$ , is the share of old capital in the total stock of capital.<sup>78</sup>

Within each of the sectors there is considerable heterogeneity in plant size, vintage, etc. Unfortunately, we are unable to incorporate such a high level of detail into this work. However, we do note that, on average, different industries emissions enter the atmosphere at different levels. Following Lvovsky and Hughes we classify emission sources as low, medium, and high height. As a first approximation, emissions from the electric power sector are classified as high height, most of the manufacturing industries are classified as medium, and the non-manufacturing and household sectors as low. The exact designations by sector are given in Table 1. Denoting the emissions of pollutant  $x$  at height  $c$  by  $E_{cxt}$  we have:

$$(3) \quad E_{cxt} = \sum_{j \in c} EM_{jxt} \quad , \quad \text{where } c = \text{low, medium, high} \quad .$$

The next step is to estimate concentrations of pollutants in population centers due to these emissions. A good approach would be to disaggregate the emissions by geographic location and feed the data into an air dispersion model at each location. This would generate the concentrations at each population center from all sources of emissions. Such an elaborate exercise will have to be deferred to future work. Again we follow Lvovsky and Hughes and use reduced form coefficients to estimate the concentrations. Unlike Lvovsky and Hughes, who distinguish between large and other cities, we make a further simplification here and express the national average urban ambient concentration as:

$$(4) \quad C_{xt}^N = \gamma_{low,x} E_{low,xt} + \gamma_{medium,x} E_{medium,xt} + \gamma_{high,x} E_{high,xt} \quad ,$$

where the  $\gamma_{cx}$  coefficients translate emissions at height  $c$  to concentration of  $x$ .<sup>79</sup>

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<sup>77</sup> For example, Jorgenson and Wilcoxon (1990) studied the economic effects of regulations in the U.S. using data on capital and operating costs of equipment that were installed in response to EPA regulations.

<sup>78</sup> This simple approach ignores the fact that cleaner equipment will likely cost more than dirty equipment. Furthermore, the exogenous energy efficiency improvements described above are set independently of these emission factors. An integrated approach would of course be preferred when such data becomes available.

<sup>79</sup> Indexing this equation by cities would be more appropriate if we had a model that calculated economic activity regionally. At a minimum we would need to have projections of population by city to make use of such a disaggregation.

The formulation described above is rather crude and so we now briefly discuss the effects of misspecification of different parts of the procedure. An error in the  $\gamma_{cx}$  reduced form coefficients has a first-order effect on the level of concentration, which as we describe next, will have a first-order effect on the estimate of health damage. This has an important direct impact on the estimates of the absolute level of the value of damages. However, when we discuss the effects of policy changes (e.g. what is the percentage reduction in mortality due to a particular policy?), then an error in  $\gamma_{cx}$  would have only a second-order effect. (In this model this parameter only enters linearly, and with no feedback, so there are no second-order effects. However, in a more general specification, there will be.) This is illustrated numerically in section six below.

Much debate and research is ongoing about the magnitude of the effect a particular level of concentration of a pollutant has on human health and on how the effects of various pollutants interact. Since much of the existing research has been done in developed countries, questions have been raised as to how these dose-response relationships should be translated to countries like China with very different pollution mixes and populations with different demographic and health characteristics. This is discussed in Wang and Smith (1999b, Appendix E) who also cite a range of estimates for mortality effects ranging from 0.04% to 0.30% for a one  $\mu g / m^3$  increase in PM-10 (see their Table 5). In addition, there is the issue of differential age impacts of these pollutants and the associated difficulty of measuring the “quality of life-years.”

We will not be able to address these important issues here and choose only a simple formulation. In our base case we follow Lvovsky and Hughes (1997) who identify eight separate health effects for PM-10 and two for SO<sub>2</sub>. The most important of these effects are mortality and chronic bronchitis. These effects, indexed by  $h$ , are given in Table 2 together with the dose-response relationship,  $DR_{hx}$ . The 7.1 number for mortality is interpreted as the number of excess deaths per million people due to an increase in the concentration of PM-10 of one  $\mu g / m^3$ . This is equivalent to a 0.1% mortality effect, which is also the central estimate in Wang and Smith (1999b, Table 5). We use an alternative estimate in our sensitivity analysis in section six.

With these dose-response relationships, the number of cases of health effect  $h$  in period  $t$  is then given by:

$$(5) \quad HE_{ht} = \sum_x (DR_{hx} (C_{xt}^N - \alpha_x) POP_t^u er) \quad h = \text{Mortality, RHA, ...},$$

where  $\alpha_x$  is the WHO reference concentration,  $POP_t^u$  is the urban population (in millions), and  $er$  is the exposure rate (the share of the urban population exposed to pollution of concentration  $C_{xt}^N$ ).

Various approaches have been used to value these damages. We use the “willingness to pay” method. The valuation of these damages is a controversial and difficult exercise, with arguments over the idea itself [Heinzerling (1999)], whether the “contingent valuation” method works [Hammit and Graham (1999)], and how to aggregate the willingness to pay [Pratt and Zeckhauser (1996)]. For this preliminary effort we again follow Lvovsky and Hughes (1997) and use estimates for willingness to pay in the U.S. and scale them by the ratio of per capita incomes in China and the U.S.<sup>80</sup> Using this simple scaling means that we are assuming a linear income effect. The U.S. values associated with each health effect are given in the third column of numbers in Table 2. The next column gives the values scaled using per capita incomes in 1995.

Most studies of health damage valuation would use these estimates for all years of their analysis. However, China is experiencing rapid increases in real incomes. For example if income rises at an annual rate of 5%, it would have risen 3.4 times in 25 years. In the base case, our model projects an average growth rate of 4-5% in per capita incomes over the next 40 years. Given this rate of increase, we have chosen a valuation method that changes every period in line with income growth, again assuming a linear income effect. The values for 2020 are given in the last column of Table 2. The national value of damage due to effect  $h$  is given by:

$$(6) \quad \text{Damage}_{ht} = \sum_x V_{ht} HE_{ht} \quad ,$$

where the valuations for 1995,  $V_{h,1995}$ , are in the third column of Table 2. The value of total damages is simply the sum over all effects:

$$(7) \quad TD_t = \sum_h \text{Damage}_{ht} \quad .$$

We should point out that these are the valuations of people who suffer the health effect. This is not the same as calculating the medical costs, the cost of lost output of sick workers, the cost of parents time to take care of sick babies, etc. The personal willingness-to-pay may, or may not, include these costs, especially in a system of publicly provided medical care.

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<sup>80</sup> These estimates are from Chapter 2 of World Bank (1997), which also discusses the use of “willingness to pay” valuation versus “human capital” valuation, the method most commonly used in China.

Table 2. Dose-Response and Valuation Estimates for PM-10 and SO<sub>2</sub>

Health Effect	Cases per 1 mil.			
	people with a 1 µg/m <sup>3</sup> increase	Valuation in 1995 U.S. \$	Valuation in 1995 yuan	Valuation in 2020 yuan
<b>Due to PM-10:</b>				
	7.14	3,600,000	82,700.00	289,000.00
2 Respiratory hospital admissions (cases)	12.00	4,750	110.00	380.00
3 Emergency room visits (cases)	235.00	140	3.20	11.20
4 Restricted activity days (days)	57,500.00	60	1.40	4.90
5 Lower respiratory infection/child asthma	23.00	50	1.10	4.00
6 Asthma attacks (cases)	2,608.00	50	1.10	4.00
7 Chronic bronchitis (cases)	61.20	72,000	1,650.00	5,770.00
8 Respiratory symptoms (cases)	183,000.00	50	1.10	4.00
<b>Due to SO<sub>2</sub>:</b>				
9	10,000.00	50	1.10	4.00
10 Respiratory systems/child	5.00	50	1.10	4.00

Sources: Dose-response data are from World Bank (1997), updated. Valuation in U.S. \$ are from Lvovsky and Hughes (1997). Valuation in yuan are author's estimates.

### 2.3 Data

Obviously, to implement the model described above, a great deal of economic and health related data is required. We need economic data for the base year, the parameters of the various behavioral functions (e.g. elasticities of substitution in the production functions), and projections of the exogenous variables. This includes projections of the population, the savings rate, productivity growth, import prices, the government deficit, etc. These data and forecasts for the economic component of the model are described in Garbaccio, Ho, and Jorgenson (1999). A particularly important data source is the 1992 Chinese input-output table.



For the health component described in section two above we obtained the output and energy use from the 1992 input-output table and Sinton (1996). The process emissions coefficients are calculated from the sectoral non-combustion emissions data in Sinton (1996). The energy related emission coefficients ( $\psi_{jxf}$ ) are derived from those in Lvovsky and Hughes (1997) and scaled to equal the combustion emissions data.<sup>81</sup> Data is given in detail for the mining, manufacturing, and electric power sectors, with summary estimates for the other sectors (agriculture, services, and final demand). We distribute the total for the other sectors in proportion to fuel use and scale Lvovsky and Hughes' estimates of these  $\sigma_{jx}$  and  $\psi_{jxf}$  coefficients. Lvovsky and Hughes also provided separate estimates of  $\psi_{jxf}^O$  and  $\psi_{jxf}^N$  and for process coefficients,  $\sigma_{jx}^O$  and  $\sigma_{jx}^N$ . The estimates for PM-10 for current and low-cost improved technology are given in Table 3a for combustion emissions and in Table 3b for process emissions.

Lvovsky and Hughes (1997) give coefficients that transform emissions to concentrations separately for each of 11 major cities. We use this information to calculate a national average set of  $\gamma_{cx}$ 's. In the 1992 base year, with emissions calibrated to the data from Sinton (1996), the estimated urban concentration averaged over the cities is  $194 \mu g / m^3$ .

Estimating the number of people affected by air pollution involves estimating and projecting the size of the urban population. Both the future total population and the urbanized portion have to be projected. We take total population projections from the World Bank (1995). The rate of urbanization in China for 1950-97 is plotted in Figure A1. For comparison we also plot the rate of urbanization in the U.S. over the period 1840-1940.<sup>82</sup> The "medium" urbanization projection is produced by letting the urbanization rate rise at 0.5% per year, while in the "low" urbanization projection, the rate is assumed to be 0.3% per year. The medium projection is very close to U.S. historical rates. Lvovsky and Hughes (1997) assume a rate of urbanization slightly higher than our medium projection.

## 2.4 *The base case simulation*

It is *not* the aim of this paper to provide estimates of the damage caused by urban air pollution, but rather the changes in the damage caused by some policy. It is only to give a clear idea of how our approach works that we describe our base case simulation, i.e. a simulation of the economy and health effects using current policy parameters.

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<sup>81</sup> Sinton (1996) provides a convenient English compilation from various Chinese sources including the China Environmental Yearbook. Page 18 gives the energy conversion coefficients. Table VIII-4 gives the emissions by sector from both combustion and noncombustion sources.

<sup>82</sup> From U.S. Census Bureau publication CPH-2-1, at <http://www.census.gov/population/censusdata/ur-def.htm>.

We start the simulation in 1995 and so we initialize the economy to have the capital stocks that were available at the start of 1995 and the working age population of 1995 supplying labor. The economic model described in the appendix calculates the output of all commodities, consumption by households and the government, exports, and the savings available for investment. This investment augments the capital stock for the next period and we repeat the exercise. The level of output (specific commodities and total GDP) thus calculated depends on our projections of the population, savings behavior, changes in spending patterns as incomes rise, the ability to borrow from abroad, improvements in technology, etc. Our results are reported in Table 4 and Figure A2. The 5.9% growth rate of GDP over the next 25 years that results from our assumptions is slightly less optimistic than the 6.7% growth rate projected recently for China by the World Bank (1997), but still implies a very rapid growth in per capita income. The population is projected to rise at a 0.7% annual rate during these 25 years.

Table 3a. **Combustion particulate emissions**

Sector	Current Emissions by Fuel			Emissions with Low Cost Improvements by Fuel		
	Coal	Oil	Natural Gas	Coal	Oil	Natural Gas
1 Agriculture	42,560	160	27	21,280	160	27
2 Coal Mining	38,182	143	24	19,091	143	24
3 Crude Petroleum	38,182	143	24	19,091	143	24
4 Metal Ore Mining	38,182	143	24	19,091	143	24
5 Other Non-metallic Ore Mining	38,182	143	24	19,091	143	24
6 Food Manufacturing	32,983	124	21	16,492	124	21
7 Textiles	18,505	69	12	9,253	69	12
8 Apparel & Leather Products	7,678	29	5	3,839	29	5
9 Lumber & Furniture Manufacturing	25,629	949	27	10,990	949	27
10 Paper, Cultural, & Educational Articles	25,629	949	27	10,990	949	27
11 Electric Power	32,642	544	0	10,881	544	0
12 Petroleum Refining	7,235	723	12	2,412	723	12
13 Chemicals	17,898	1,790	30	5,966	1,790	30
14 Building Material	13,454	1,345	22	4,485	1,345	22
15 Primary Metals	6,379	638	11	2,126	638	11
16 Metal Products	8,814	33	6	4,407	33	6
17 Machinery	11,970	45	7	5,985	45	7
18 Transport Equipment	11,970	45	7	5,985	45	7
19 Electric Machinery & Instruments	11,970	45	7	5,985	45	7
20 Electronic & Communication Equipment	11,970	45	7	5,985	45	7
21 Instruments and Meters	11,970	45	7	5,985	45	7
22 Other Industry	46,872	176	29	23,436	176	29
23 Construction	42,560	160	27	21,280	160	27
24 Transportation & Communications	42,560	5,320	27	21,280	2,660	27
25 Commerce	42,560	160	27	21,280	160	27
26 Public Utilities	42,560	160	27	21,280	160	27
27 Culture, Education, Health, & Research	42,560	160	27	21,280	160	27
28 Finance & Insurance	42,560	160	27	21,280	160	27
29 Public Administration	42,560	160	27	21,280	160	27
Households	21,280	426	27	10,640	426	27

Note: Coefficients  $\psi_{jxf}^O$  and  $\psi_{jxf}^N$  in tons of PM-10 per million tons of oil equivalent (toe).

Table 3b. **Process Particulate Emissions**

Sector	Current Emissions	Emissions with Low Cost Improvements
1 Agriculture	-	-
2 Coal Mining	0.81	0.16
3 Crude Petroleum	0.81	0.16
4 Metal Ore Mining	0.81	0.16
5 Other Non-metallic Ore Mining	0.81	0.16
6 Food Manufacturing	0.09	0.09
7 Textiles	0.04	0.04
8 Apparel & Leather Products	--	--
9 Lumber & Furniture Manufacturing	0.12	0.02
10 Paper, Cultural, & Educational Articles	0.12	0.02
11 Electric Power	0.72	0.72
12 Petroleum Refining	0.57	0.57
13 Chemicals	0.71	0.71
14 Building Material	14.92	2.98
15 Primary Metals	3.17	0.63
16 Metal Products	0.05	0.05
17 Machinery	0.11	0.11
18 Transport Equipment	0.11	0.11
19 Electric Machinery & Instruments	0.11	0.11
20 Electronic & Communication Equipment	0.11	0.11
21 Instruments and Meters	0.11	0.11
22 Other Industry	1.53	1.53
23 Construction	--	--
24 Transportation & Communications	--	--
25 Commerce	--	--
26 Public Utilities	--	--
27 Culture, Education, Health, & Research	--	--
28 Finance & Insurance	--	--
29 Public Administration	--	--
Households	--	--

Note: Coefficients  $\sigma_{jx}^O$  and  $\sigma_{jx}^N$  in tons per million 1992 yuan.

Table 4. Selected Variables from Base Case Simulation

Variable	1995	2010	2030
Population (mil.)	1,200.00	1,348.00	1,500.00
GDP (bil. 1992 <i>yuan</i> )	3,560.00	10,200.00	18,600.00
Energy Use (mil. tons sce)	1,190.00	2,490.00	3,280.00
Coal Use (mil. tons)	1,270.00	2,580.00	3,090.00
Oil Use (mil. tons)	180.00	420.00	690.00
Carbon Emissions (mil. tons)	810.00	1,670.00	2,160.00
Particulate Emissions (mil. tons)	21.55	26.78	33.84
From High Height Sources	3.94	4.81	6.80
From Medium Height Sources	11.30	12.48	15.99
From Low Height Sources	6.32	9.49	11.05
SO <sub>2</sub> Emissions (mil. tons)	21.80	42.40	57.90
Premature Deaths (1,000)	320.00	700.00	1,200.00
Health Damage (bil. <i>yuan</i> )	180.00	1,000.00	2,800.00
Health Damage/GDP	5.10%	9.80%	15.30%

The dashed line in Figure A2 shows the fossil fuel based energy use in standard coal equivalents (sce) on the right-hand axis. Our assumptions on energy use improvements are fairly optimistic and together with changes in the structure of the economy, result in an energy-GDP ratio in 2030 that is almost half that in 1995. The carbon emissions from fossil fuels are also plotted using the right-hand axis. The rate of growth of carbon emissions is even slower than the growth in energy use. This is mainly due to our assumptions on the shift from coal to oil.

With the industry outputs and input requirements calculated for each period we use equations (1)-(7) to calculate total emission of pollutants, the urban concentration of pollutants, and the health effects of these pollutants. The growth of PM-10 emissions is much slower than the growth in energy use and carbon emissions. This is due to the sharp difference in the assumed coefficients for new and old capital (see Table 3). All sources of PM-10 increase emissions, with the largest rise coming from low-height sources. Projected SO<sub>2</sub> emissions rise much faster than particulates due to a less optimistic estimate of the improvement in the  $\sigma_{jx}$  and  $\psi_{jxf}$  coefficients.<sup>83</sup>

In this base case we assume no increase in emission reduction efforts over time. This differs from Lvovsky and Hughes' (1997) BAU case which assumes that the largest 11 cities will choose what they call the "high investment" option. The result is that our estimate of current premature mortality is higher, 320,000 versus 230,000. The growth rate of health effects from our simulations, however, are quite close. By 2020 our estimated excess deaths are 3.1 times the 1995 level, compared to the 3.7 times calculated in Lvovsky and Hughes' BAU case.

Of course the fact that our estimates are close does not mean that either estimate is "good." We report the level estimates to explain our simulation procedure and to illustrate the magnitudes involved. To reiterate, this is not a forecast of emissions, but rather a projection if no changes in policy are made. We expect both the government and private sectors to have policies and investments that are different from today's. The important issue is policy choices and the estimation of the effects of different policies. This is where we turn next.

## 2.5 *Health effects of a carbon tax*

As described in the previous section, our projected growth of carbon emissions in the base case, while lower than the growth of GDP, is still very high. The level of emissions doubles in 15 years. A number of policies have been suggested to reduce the growth of emissions of this global pollutant, ranging from specific, detailed policies like importing natural gas or shutting down small coal plants, to broader approaches, such as carbon taxes and emissions trading. In this paper we concentrate on the simplest broad based policy by imposing a "carbon tax," i.e. a tax on fossil fuels based on their carbon content.<sup>84</sup>

The specifics of this tax, and the detailed economic effects, are discussed in Garbaccio, Ho, and Jorgenson (1999). In our simulations we raise the price of crude petroleum and coal, both domestic and imported, by this carbon tax. In this paper, two carbon targets are examined, 5% and 10% reductions in annual carbon emissions. The level of the tax is calculated endogenously such that emissions in each period are 5% or 10% less than in the base case. This is shown in Figure 1. The revenues from this new tax are used to reduce other existing taxes. The amount of reduction is such that the public deficit (exogenous) and real government expenditures (endogenous) were kept the same as the base case.

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<sup>83</sup> The emission coefficients for sulphur dioxide are not reported here but are available from the authors. Given the relatively minor role in human health (as shown in Table 2), we do not emphasize SO<sub>2</sub> in this study. It is of course an important cause of other damages, e.g. acid rain.

<sup>84</sup> In this study we ignore both other sources of carbon dioxide and other greenhouse gases.

The results of these carbon tax simulations are given in Table 5 and Figures 1-4. The amount of carbon tax needed to achieve these reductions is plotted in Figure 2. In the first year, a tax of 8.8 *yuan* per ton is required to achieve a 5% reduction in emissions.<sup>85</sup> This is equivalent to a 6% increase in the factory gate price of coal and a 1% increase in the price of crude petroleum. These higher energy prices reduce demand for fuels and raise the relative prices of energy intensive goods. We assume that the government does not compensate the household sector for the higher prices and so consumption falls in the short run. Because the labor supply is assumed fixed, real wages fall slightly. The compensating reduction in enterprise taxes, however, leaves firms with higher after-tax income, and given our specification, this leads to higher investment. Over time, this leads to a significantly higher capital stock, i.e., higher than in the base case, and thus higher GDP. This higher output allows a level of consumption that exceeds that in the base case soon after the beginning of the simulation period.

As can be seen in Table 5, in the first year of the 5% carbon reduction case, the imposition of the carbon tax leads to a reduction in total particulate emissions of 3.5%. This, however, is an average over three different changes. High height emissions from the electric power sector fell by 5.6%, medium height emissions from manufacturing fell 2.7%, while low height emissions fell 3.7%. Sectoral emissions of sulfur dioxide fell by similar amounts. The electric power sector is the most fossil fuel intensive and hence experiences the largest fall in output and emissions.

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<sup>85</sup> There are 0.518 tons of carbon emitted per ton of average coal. The average price of coal output in 1992, derived by dividing the value in the input-output table by the quantity of coal mined, is about 68 *yuan* per ton. This implies a tax on coal of about 7 percent.

Table 5. **Effects of a Carbon Tax on Selected Variables**  
(Percentage Change from Base Case)

Variable	Effect in 1st Year with:		Effect in 15th Year with:	
	5% CO <sub>2</sub> Emissions Reduction	10% CO <sub>2</sub> Emissions Reduction	5% CO <sub>2</sub> Emissions Reduction	10% CO <sub>2</sub> Emissions Reduction
GDP	-0.00%	-0.00%	0.21%	0.42%
Primary Energy	-4.72%	-9.45%	-4.68%	-9.35%
Market Price of Coal	6.03%	12.80%	6.29%	13.40%
Market Price of Oil	0.95%	2.01%	0.71%	1.53%
Coal Output	-5.93%	-11.80%	-6.14%	-12.20%
Oil Output	-0.81%	-1.71%	-0.59%	-1.29%
Particulate Emissions	-3.50%	-6.97%	-3.11%	-6.20%
From High Height Sources	-3.67%	-7.37%	-3.09%	-6.21%
From Medium Height Sources	-2.66%	-5.29%	-2.22%	-4.40%
From Low Height Sources	-5.59%	-11.20%	-5.44%	-10.90%
Particulate Concentration	-3.45%	-6.92%	-2.95%	-5.92%
SO <sub>2</sub> Concentration	-3.43%	-6.88%	-2.78%	-5.59%
Premature Deaths	-4.52%	-9.04%	-3.55%	-7.10%
Cases of Chronic Bronchitis	-4.52%	-9.04%	-3.55%	-7.10%
Value of Health Damages	-4.52%	-9.04%	-3.55%	-7.11%



Figure 1. Carbon emissions in base case and simulations

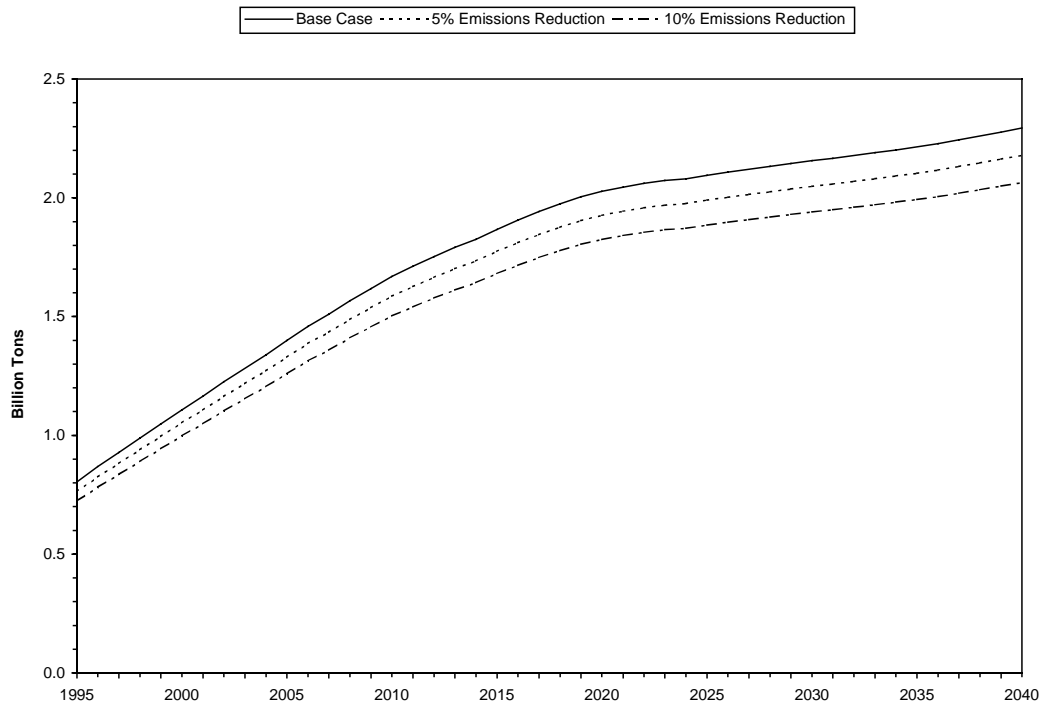


Figure 2. Carbon taxes required to attain a given reduction in emissions

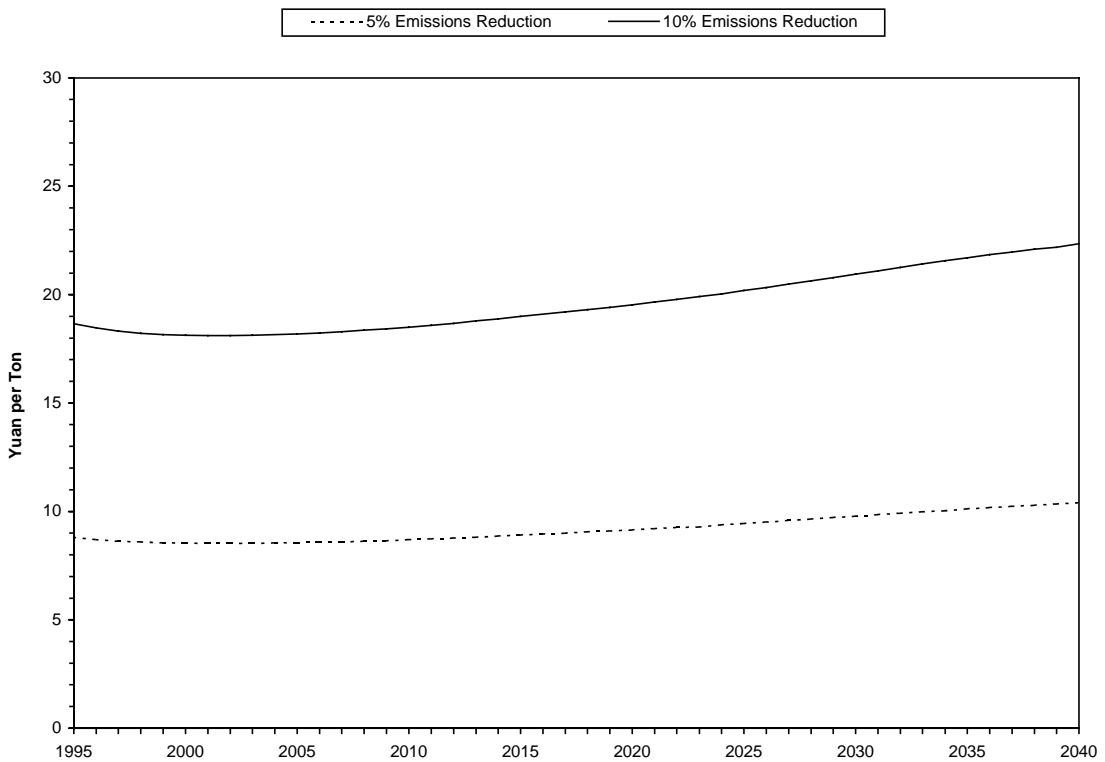


Figure 3. Reduction in PM-10 emissions and concentrations relative to base

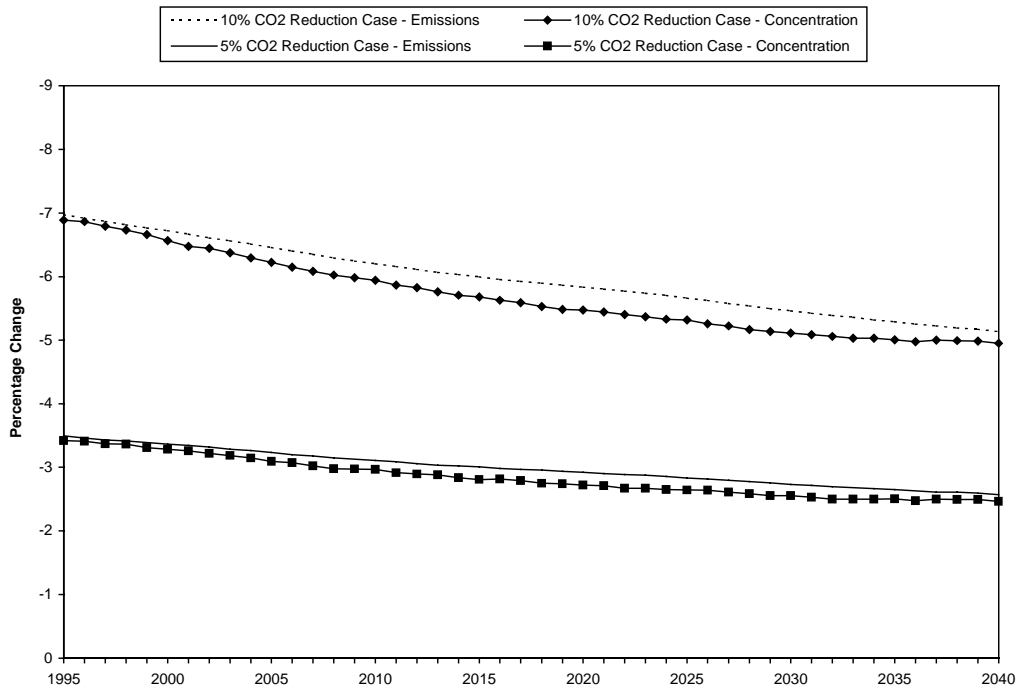
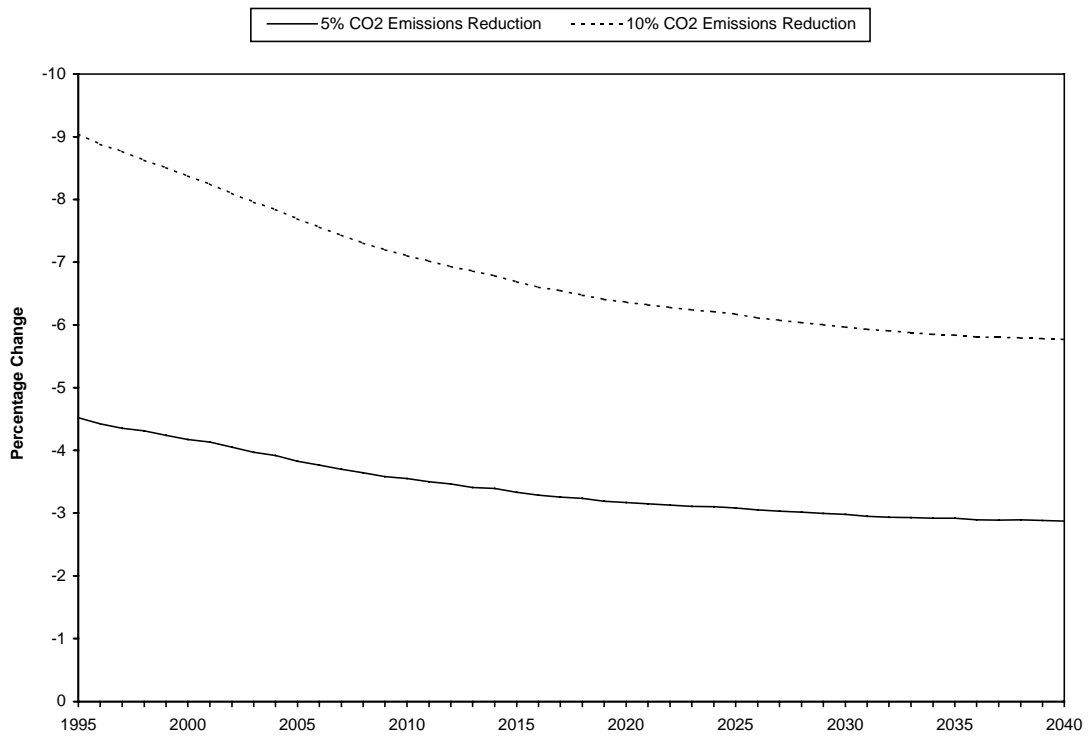


Figure 4. Reduction in excess deaths relative to base case



This reduction in emissions results in a fall in the average urban concentration of PM-10 by 3.4%. As a consequence, cases of various health effects fall by about 4.5% (i.e. the number of premature deaths, the number of cases of chronic bronchitis, etc.). The reduction in health effects is bigger than the change in concentration due to the non-proportional nature of equation 5. If we apply these percent changes to the base case estimates in Table 4, this translates to 14,000 fewer excess deaths, and 126,000 fewer cases of chronic bronchitis. Since the valuations are simple multiples (see equation 6) the percent change in *yuan* values of this health damage is also -4.5%.

Over time, as the revenue raised through the carbon tax reduces the income tax burden on enterprises, higher investment leads to a larger capital stock and hence a higher level of GDP. The higher level of output means greater demand for energy and hence requires a higher carbon tax rate to achieve the 5% reduction in carbon emissions. This is shown in the “15th year” column of Table 5 and in Figure 2. The lower tax on crude petroleum in the 15th year is due to our assumption on the price of world oil. If we had assumed no imports, the tax on crude petroleum would also have been higher. This twist in fossil fuel prices results in a bigger fall in coal consumption compared to crude petroleum consumption for an unchanged GDP. However, the higher demand has a bigger effect than this twist in fuel prices and hence the reduction in emissions in the 15th year is smaller than the initial reduction, 3.1% versus 3.5%. The reduction in concentrations over time is correspondingly smaller, as shown in Figure 3.

Another feature of the results that should be pointed out is that the change in concentration is smaller than the change in emissions in future years (see Figure 3). This is due to our classifying emissions by height and that low level emissions are the biggest contributors to concentration (i.e. the biggest  $\gamma_{cx}$  ‘s). Different sectors of the economy are growing at different rates (sources of low height emissions are growing the most rapidly), and respond differently to the imposition of the carbon tax. The most responsive sector (i.e. the one that shrinks the most) is electric power generation, which produces high height emissions with the lowest contribution to concentrations. Finally, this path of concentration changes leads to health effects that become smaller over time, from a 4.5% reduction in the first year, to a 3.6% reduction in the 15th year, to 3.2% in the 25th year.

When we raise the targeted carbon emissions reductions from 5% to 10% of the base case the effects are approximately linear. In the “15th year” column of Table 5 we see that the effects on coal prices are less than doubled while the effects on oil prices are more than doubled. The end result on emissions, concentrations, and health effects is a simple doubling of the percentage change. This seeming linearity would not hold for larger changes.

## 2.6 Sensitivity analysis

In section three above we discuss first- and second-order effects of an error in a parameter. To illustrate this we use an alternative assumption about an exogenous variable, the future urbanization rate. This variable,  $POP_t^u$ , enters in equation 4. The base case plotted in Figure A1 has the urban share of total population rising at 0.5% per year, the “low” case rises at 0.3% per year. We ran the model again with this lower estimate of the exposed population. The number of premature deaths in both the base case and in this alternative simulation are plotted in Figure A3. This is an example of a first-order effect of an error in a parameter or exogenous variable.

Finally, we ran the model with the lower urban population growth rate and again imposed a carbon tax to achieve a 5% reduction in carbon emissions. In the original simulation, this resulted in excess deaths that were 4.5% lower in the first year (see Table 5 and Figure 4). Using the alternative urbanization estimate, mortality again falls by 4.5%. The *percentage* reductions in premature mortality over time for both cases are plotted in Figure A4. They are almost identical.

The wide range of estimates for the dose-response relationship was noted in section three above. For mortality, Lvovsky and Hughes (1997) use 7.14 excess deaths per million per  $\mu g / m^3$ . If we use a coefficient that is 1.5 times higher, well within the range cited by Wang and Smith (1999b), then the projected excess deaths are simply 50% higher. This is shown in Figure A5.

The above illustrates the effect of changing a variable or parameter that has no feedback effect. However, if we change exogenous variables that do have feedbacks, there will be second-order effects. For example, an alternative guess about the time path of the government deficit will change revenue requirements and taxes and will have an effect on the estimated percentage change. This effect will, however, be minor, merely a second-order effect on the percentage change.

The really crucial parameters have a first-order effect on the percentage change. These include the elasticity of substitution between capital and energy, the elasticity of substitution between coal and oil, and other behavioral parameters. In the case of health effects, if the concentration and dose responses (equations 4 and 5) were not linear then there would be significant changes. Two other examples come to mind. If the health of workers is a factor in the effectiveness of labor input or if urbanization is modeled explicitly, then something like a carbon tax would have a more complex interaction with GDP and health benefits. Examination of these issues is deferred to future studies.

## 2.7 *Conclusions*

This paper presents a preliminary effort to integrate a model of health effects from fossil fuel use with a multi-sector economy-wide CGE model. In our initial analysis, we look at how policies intended to reduce emissions of greenhouse gasses might simultaneously affect emissions of local pollutants and ultimately human health. Our initial specifications of the linkages between fuel use and emissions of local pollutants and between emissions of these pollutants and their concentrations in urban areas are very simple. Efforts to improve these specifications are currently under way. However, to the extent that the effects are linear (as described in equations 4 and 5), our estimates of the percent changes in concentrations and mortality would be as good (or bad) as our estimates of sectoral output changes.

The aim of a more detailed modeling effort would be to provide guidance for policy making. One goal of this preliminary effort is to lay out explicitly the assumptions that need to go into making such an analysis, even with better data and more elaborate model specification. A complex regional air pollution model would still require that projections be made about total and urban population size, future world oil prices, energy efficiency improvements, and all the other time-dependent exogenous variables discussed previously.

Another goal of this preliminary modelling effort is to highlight in which areas improvements in data collection and modelling would bring the greatest benefit to even a limited analysis. Issues beyond those associated with the economic part of the model include: (i) Health damage from air pollution is believed to be due to very fine particles. Data on that would be important. (ii) Data on concentrations in different urban areas and the modelling of these concentrations in a sample of cities would give a sense of the range of the reduced form coefficients. (iii) We have crudely classified emissions by low, medium, and high heights for different industries. Having more refined data on industry emissions characteristics would improve the modelling in item (ii). (iv) Getting better dose-response functions is already a recognized priority. We would urge consideration of including an age dimension in the research. This would be especially important in attempting to link worker's health back to labor productivity.

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## **APPENDIX A: DESCRIPTION OF THE ECONOMIC MODEL**

The main features of the model for China are discussed in this appendix, further details are given in Garbaccio, Ho, and Jorgenson (1997). We describe the modeling of each of the main agents in the model in turn. Table A1 lists a number of parameters and variables which are referred to with some frequency. In general, a bar above a symbol indicates that it is a plan parameter or variable while a tilde indicates a market variable. Symbols without markings are total quantities or average prices. To reduce unnecessary notation, whenever possible, we drop the time subscript,  $t$ , from our equations.

Table 1A. **Selected Parameters and Variables in the Economic Model**

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**Parameters**

$s_i^e$	export subsidy rate on good $i$
$t_i^c$	carbon tax rate on good $i$
$t^k$	tax rate on capital income
$t^L$	tax rate on labor income
$t_i^r$	net import tariff rate on good $i$
$t_i^t$	net indirect tax (output tax less subsidy) rate on good $i$
$t^x$	unit tax per ton of carbon

**Endogenous Variables**

$G_I$	interest on government bonds paid to households
$G_{INV}$	investment through the government budget
$G_{IR}$	interest on government bonds paid to the rest of the world
$G_{transfer}$	government transfer payments to households
$P_i^{KD}$	rental price of market capital by sector
$PE_i^*$	export price in foreign currency for good $i$
$PI_i$	producer price of good $i$
$PI_i^t$	purchaser price of good $i$ including taxes
$PL$	average wage
$PL_i$	wage in sector $i$
$PM_i$	import price in domestic currency for good $i$
$PM_i^*$	import price in foreign currency for good $i$
$PS_i$	supply price of good $i$
$PT_i$	rental price of land of type $i$
$QI_i$	total output for sector $i$
$QS_i$	total supply for sector $i$
$r(B^*)$	payments by enterprises to the rest of the world
$R_{transfer}$	transfers to households from the rest of the world

---



## A.1 Production

Each of the 29 industries is assumed to produce its output using a constant returns to scale technology. For each sector  $j$  this can be expressed as:

$$(A1) \quad QI_j = f(KD_j, LD_j, TD_j, A_{1j}, \dots, A_{nj}, t) \quad ,$$

where  $KD_j$ ,  $LD_j$ ,  $TD_j$ , and  $A_{ij}$  are capital, labor, land, and intermediate inputs, respectively.<sup>86</sup>

In sectors for which both plan and market allocation exists, output is made up of two components, the plan quota output ( $\bar{QI}_j$ ) and the output sold on the market ( $\tilde{QI}_j$ ). The plan quota output is sold at the state-set price ( $\bar{PI}_j$ ) while the output in excess of the quota is sold at the market price ( $\tilde{PI}_j$ ).

A more detailed discussion of how this plan-market formulation is different from standard market economy models is given in Garbaccio, Ho, and Jorgenson (1999). In summary, if the constraints are not binding, then the “two-tier plan/market” economy operates at the margin as a market economy with lump sum transfers between agents. The return to the owners of fixed capital in sector  $j$  is:

$$(A2) \quad \begin{aligned} profit_j = & \bar{PI}_j \bar{QI}_j + \tilde{PI}_j \tilde{QI}_j - P_j^{KD} \tilde{KD}_j - PL_j LD_j - PT_j TD_j \\ & - \sum_i \bar{PS}_i \bar{A}_{ij} - \sum_i \tilde{PS}_i \tilde{A}_{ij} \quad . \end{aligned}$$

For each industry, given the capital stock  $\bar{K}_j$  and prices, the first order conditions from maximizing equation A2, subject to equation A1, determine the market and total input demands.

Given the lack of a consistent time-series data set, in this version of the model, we use Cobb-Douglas production functions. Equation A1 for the output of industry  $j$  at time  $t$  then becomes:

$$(A3) \quad QI_{jt} = g(t) KD_{jt}^{\alpha_{kj}} LD_{jt}^{\alpha_{lj}} TD_{jt}^{\alpha_{tj}} E_{jt}^{\alpha_{Ej}} M_{jt}^{\alpha_{Mj}} \quad , \quad \text{where}$$

$$\log E_{jt} = \sum_k \alpha_{kj}^E \log A_{kjt} \quad \text{and} \quad k = \text{coal, oil, electricity, and refined petroleum} \quad ,$$

$$\log M_{jt} = \sum_k \alpha_{kj}^M \log A_{kjt} \quad \text{and} \quad k = \text{non-energy intermediate goods} \quad .$$

---

<sup>86</sup>  $QI_j$  denotes the quantity of industry  $j$ 's output. This is to distinguish it from,  $QC_j$ , the quantity of commodity  $j$ . In the actual model each industry may produce more than one commodity and each commodity may be produced by more than one industry. In the language of the input output tables, we make use of both the USE and MAKE matrices. For ease of exposition we ignore this distinction here.

Here  $\alpha_{Ej}$  is the cost share of aggregate energy inputs in the production process and  $\alpha_{kj}^E$  is the share of energy of type  $k$  within the aggregate energy input. Similarly,  $\alpha_{Mj}$  is the cost share of aggregate non-energy intermediate inputs and  $\alpha_{kj}^M$  is the share of intermediate non-energy input of type  $k$  within the aggregate non-energy intermediate input.

To allow for biased technical change, the  $\alpha_{Ej}$  coefficients are indexed by time and are updated exogenously. We set  $\alpha_{Ej}$  to fall gradually over the next 40 years while the labor coefficient,  $\alpha_{Lj}$ , rises correspondingly. The composition of the aggregate energy input (i.e. the coefficients  $\alpha_{kj}^E$ ) are also allowed to change over time. These coefficients are adjusted gradually so that they come close to resembling the U.S. use patterns of 1992. The exception is that the Chinese coefficients for coal for most industries will not vanish as they have in the U.S.<sup>87</sup> The coefficient  $g(t)$  in equation A3 represents technical progress and the change in  $g(t)$  is determined through an exponential function ( $\dot{g}_j(t) = A_j \exp(-\mu_j t)$ ). This implies technical change that is rapid initially, but gradually declines toward zero. The price to buyers of this output includes the indirect tax on output and the carbon tax:

$$(A4) \quad PI_i^t = (1 + t_i^t)PI_i + t_i^c \quad .$$

## A.2 Households

The household sector derives utility from the consumption of commodities, is assumed to supply labor inelastically, and owns a share of the capital stock. It also receives income transfers and interest on its holdings of public debt. Private income after taxes and the payment of various non-tax fees ( $FEE$ ),  $Y^p$ , can then be written as:

$$(A5) \quad Y^p = YL + DIV + G\_I + G\_transfer + R\_transfer - FEE \quad ,$$

where  $YL$  denotes labor income from supplying  $LS$  units of effective labor, less income taxes.  $YL$  is equal to:

$$(A6) \quad YL = (1 - t^L)PLLS \quad .$$

---

<sup>87</sup> We have chosen to use U.S. patterns in our projections of these exogenous parameters because they seem to be a reasonable anchor. While it is unlikely that China's economy in 2032 will mirror the U.S. economy of 1992, it is also unlikely to closely resemble any other economy. Other projections, such as those by the World Bank (1994), use the input-output tables of developed countries including the U.S. We have considered making extrapolations based on recent Chinese input-output tables, but given the short sample period and magnitude of the changes in recent years, this did not seem sensible.

The relationship between labor demand and supply is given in equation A31 below.  $LS$  is a function of the working age population, average annual hours, and an index of labor quality:

$$(A7) \quad LS_t = POP_t^w hr_t q_t^L \quad .$$

Household income is allocated between consumption ( $VCC_t$ ) and savings. In this version of the model we use a simple Solow growth model formulation with an exogenous savings rate ( $s_t$ ) to determine private savings ( $S_t^p$ ):

$$(A8) \quad S_t^p = s_t Y_t^p = Y_t^p - VCC_t \quad .$$

Household utility is a function of the consumption of goods such that:

$$(A9) \quad U_t = U(C_{1t}, \dots, C_{nt}) = \sum_i \alpha_{it}^C \log C_{it} \quad .$$

Assuming that the plan constraints are not binding, then as in the producer problem above, given market prices and total expenditures, the first order conditions derived from equation A9 determine household demand for commodities,  $C_i$ , where  $C_i = \bar{C}_i + \tilde{C}_i$ . Here  $\bar{C}_i$  and  $\tilde{C}_i$  are household purchases of commodities at state-set and market prices. The household budget can be written as:

$$(A10) \quad VCC = \sum_i (\tilde{P}S_i \tilde{C}_i + \bar{P}S_i \bar{C}_i) \quad .$$

We use a Cobb-Douglas utility function because we currently lack the disaggregated data to estimate an income elastic functional form. However, one would expect demand patterns to change with rising incomes and this is implemented by allowing the  $\alpha_{it}^C$  coefficients to change over time. These future demand patterns are projected using the U.S. use patterns of 1992.

### A.3 Government and taxes

In the model, the government has two major roles. First, it sets plan prices and output quotas and allocates investment funds. Second, it imposes taxes, purchases commodities, and redistributes resources. Public revenue comes from direct taxes on capital and labor, indirect taxes on output, tariffs on imports, the carbon tax, and other non-tax receipts:

$$(A11) \quad Rev = \sum_j t^k (P_j^{KD} KD_j - D_j) + t^L \sum_j PL_j LD_j + \sum_j t_j^t PI_j QI_j + \sum_i t_i^r PM_i^* M_i \\ + \sum_i t_i^c (QI_i - X_i + M_i) + FEE \quad ,$$

where  $D_j$  is the depreciation allowance and  $X_i$  and  $M_i$  are the exports and imports of good  $i$ . The carbon tax per unit of fuel  $i$  is:

$$(A12) \quad t_i^c = t^x \theta_i \quad ,$$

where  $t^x$  is the unit carbon tax calculated per ton of carbon and  $\theta_i$  is the emissions coefficient for each fuel type  $i$ .

Total government expenditure is the sum of commodity purchases and other payments:

$$(A13) \quad Expend = VGG + G\_INV + \sum_i s_i^e PI_i X_i + G\_I + G\_IR + G\_transfer$$

Government purchases of specific commodities are allocated as shares of the total value of government expenditures,  $VGG$ . For good  $i$ :

$$(A14) \quad PS_i G_i = \alpha_i^G VGG \quad .$$

We construct a price index for government purchases as  $\log PGG = \sum_i \alpha_i^G \log PS_i$ . The real quantity of government purchases is then:

$$(A15) \quad GG = \frac{VGG}{PGG} \quad .$$

The difference between revenue and expenditure is the deficit,  $\Delta G$ , which is covered by increases in the public debt, both domestic ( $B$ ) and foreign ( $B^{G*}$ ):

$$(A16) \quad \Delta G_t = Expend_t - Rev_t \quad ,$$

$$(A17) \quad B_t + B_t^{G*} = B_{t-1} + B_{t-1}^{G*} + \Delta G_t \quad .$$

The deficit and interest payments are set exogenously and equation A16 is satisfied by making the level of total government expenditure on goods,  $VGG$ , endogenous.

#### A.4 Capital, investment, and the financial system

We model the structure of investment in a fairly simple manner. In the Chinese economy, some state-owned enterprises receive investment funds directly from the state budget and are allocated credit on favorable terms through the state-owned banking system. Non-state enterprises get a negligible share of state investment funds and must borrow at what are close to competitive interest rates. There is also a small but growing stock market that provides an alternative channel for private savings. We abstract from these features and define the capital stock in each sector  $j$  as the sum of two parts, which we call plan and market capital:

$$(A18) \quad K_{jt} = \bar{K}_{jt} + \tilde{K}_{jt} \quad .$$

The plan portion evolves with plan investment and depreciation:

$$(A19) \quad \bar{K}_{jt} = (1 - \delta) \bar{K}_{jt-1} + \bar{I}_{jt} \quad , \quad t = 1, 2, \dots, T .$$

In this formulation,  $\bar{K}_{j0}$  is the capital stock in sector  $j$  at the beginning of the simulation. This portion is assumed to be immobile across sectors. Over time, with depreciation and limited government investment, it will decline in importance. Each sector may also “rent” capital from the total stock of market capital,  $\tilde{K}_t$ :

$$(A20) \quad \tilde{K}_t = \sum_j \tilde{K}_{jt} \quad , \quad \text{where} \quad \tilde{K}_{jt} > 0 \quad .$$

The allocation of market capital to individual sectors,  $\tilde{K}_{jt}$ , is based on sectoral rates of return. As in equation A2, the rental price of market capital by sector is  $\tilde{P}_j^{KD}$ . The supply of  $\tilde{K}_{jt}$ , subject to equation A20, is written as a translog function of all of the market capital rental prices,  $\tilde{K}_{jt} = K_j(\tilde{P}_1^{KD}, \dots, \tilde{P}_n^{KD})$ .

In two sectors, agriculture and crude petroleum, “land” is a factor of production. We have assumed that agricultural land and oil fields are supplied inelastically, abstracting from the complex property rights issues regarding land in China. After taxes, income derived from plan capital, market capital, and land is either kept as retained earnings by the enterprises, distributed as dividends, or paid to foreign owners:

$$(A21) \quad \sum_j profits_j + \sum_j \tilde{P}_j^{KD} \tilde{K}_j + \sum_j PT_j T_j = tax(k) + RE + DIV + r(B^*) \quad ,$$

where  $tax(k)$  is total direct taxes on capital (the first term on the right hand side of equation A11).<sup>88</sup>

As discussed below, total investment in the model is determined by savings. This total,  $VII$ , is then distributed to the individual investment goods sectors through fixed shares,  $\alpha_{it}^I$ :

$$(A22) \quad PS_{it} I_{it} = \alpha_{it}^I VII_t \quad .$$

Like the  $\alpha_{it}^C$  coefficients in the consumption function, the investment coefficients are indexed by time and projected using U.S. patterns for 1992. A portion of sectoral investment,  $\bar{I}_t$ , is allocated directly by the government, while the remainder,  $\tilde{I}_t$ , is allocated through other channels.<sup>89</sup> The total,  $I_t$ , can be written as:

<sup>88</sup> In China, most of the “dividends” are actually income due to agricultural land.

<sup>89</sup> It should be noted that the industries in the Chinese accounts include many sectors that would be considered public goods in other countries. Examples include local transit, education, and health.

$$(A23) \quad I_t = \tilde{I}_t + \bar{I}_t = I_{1t}^{\alpha_1^I} I_{2t}^{\alpha_2^I} \dots I_{nt}^{\alpha_n^I} .$$

As in equation A19 for the plan capital stock, the market capital stock,  $\tilde{K}_{jt}$ , evolves with new market investment:

$$(A24) \quad \tilde{K}_{jt} = (1 - \delta) \tilde{K}_{jt-1} + \tilde{I}_{jt} .$$

## A.5 The foreign sector

Trade flows are modeled using the method followed in most single-country models. Imports are considered to be imperfect substitutes for domestic commodities and exports face a downward sloping demand curve. We write the total supply of commodity  $i$  as a CES function of the domestic ( $QI_i$ ) and imported good ( $M_i$ ):

$$(A25) \quad QS_i = A_0 [\alpha^d QI_i^\rho + \alpha^m M_i^\rho]^{\frac{1}{\rho}} ,$$

where  $PS_i QS_i = PI_i^d QI_i + PM_i M_i$  is the value of total supply. The purchaser's price for domestic goods,  $PI_i^d$ , is discussed in the producer section above. The price of imports to buyers is the foreign price plus tariffs (less export subsidies), multiplied by a world relative price,  $e$ :

$$(A26) \quad PM_i = e(1 + t_i^r) PM_i^* .$$

Exports are written as a simple function of the domestic price relative to world prices adjusted for export subsidies ( $s_{it}^e$ ):

$$(A27) \quad X_{it} = EX_{it} \left( \frac{\tilde{PI}_{it}}{e_t (1 + s_{it}^e) PE_{it}^*} \right)^{\eta} ,$$

where  $EX_{it}$  is base case exports that are projected exogenously.

The current account balance is equal to exports minus imports, less net factor payments, plus transfers:

$$(A28) \quad CA = \sum_i \frac{PI_i X_i}{(1 + s_i^e)} - \sum_i PM_i M_i - r(B^*) - G_{IR} + R_{transfer} ,$$

Like the government deficits, the current account balances are set exogenously and accumulate into stocks of net foreign debt, both private ( $B_t^*$ ) and public ( $B_t^{G^*}$ ):

$$(A29) \quad B_t^* + B_t^{G*} = B_{t-1}^* + B_{t-1}^{G*} - CA_t \quad .$$

## A.6 Markets

The economy is in equilibrium in period  $t$  when the market prices clear the markets for the 29 commodities and the two factors. The supply of commodity  $i$  must satisfy the total of intermediate and final demands:

$$(A30) \quad QS_i = \sum_j A_{ij} + C_i + I_i + G_i + X_i \quad , \quad i = 1, 2, \dots, 29.$$

For the labor market, we assume that labor is perfectly mobile across sectors so there is one average market wage which balances supply and demand. As is standard in models of this type, we reconcile this wage with the observed spread of sectoral wages using wage distribution coefficients,  $\psi_{jt}^L$ . Each industry pays  $PL_{jt} = \psi_{jt}^L PL_t$  for a unit of labor. The labor market equilibrium is then given as:

$$(A31) \quad \sum_j \psi_{jt}^L LD_{jt} = LS_t \quad .$$

For the non-plan portion of the capital market, adjustments in the market price of capital,  $\tilde{P}_j^{KD}$ , clears the market in sector  $j$ :

$$(A32) \quad KD_{jt} = \psi_{jt}^K K_{jt} \quad ,$$

where  $\psi_{jt}^K$  converts the units of capital stock into the units used in the production function. The rental price  $PT_j$  adjusts to clear the market for “land”:

$$(A33) \quad TD_j = T_j \quad , \quad \text{where } j = \text{“agriculture” and “petroleum extraction.”}$$

In this model without foresight, investment equals savings. There is no market where the supply of savings is equated to the demand for investment. The sum of savings by households, businesses (as retained earnings), and the government is equal to the total value of investment plus the budget deficit and net foreign investment:

$$(A34) \quad S^p + RE + G\_INV = VII + \Delta G + CA \quad .$$

The budget deficit and current account balance are fixed exogenously in each period. The world relative price ( $e$ ) adjusts to hold the current account balance at its exogenously determined level.

Figure A1. Data and projections of Urban population as a percentage of total

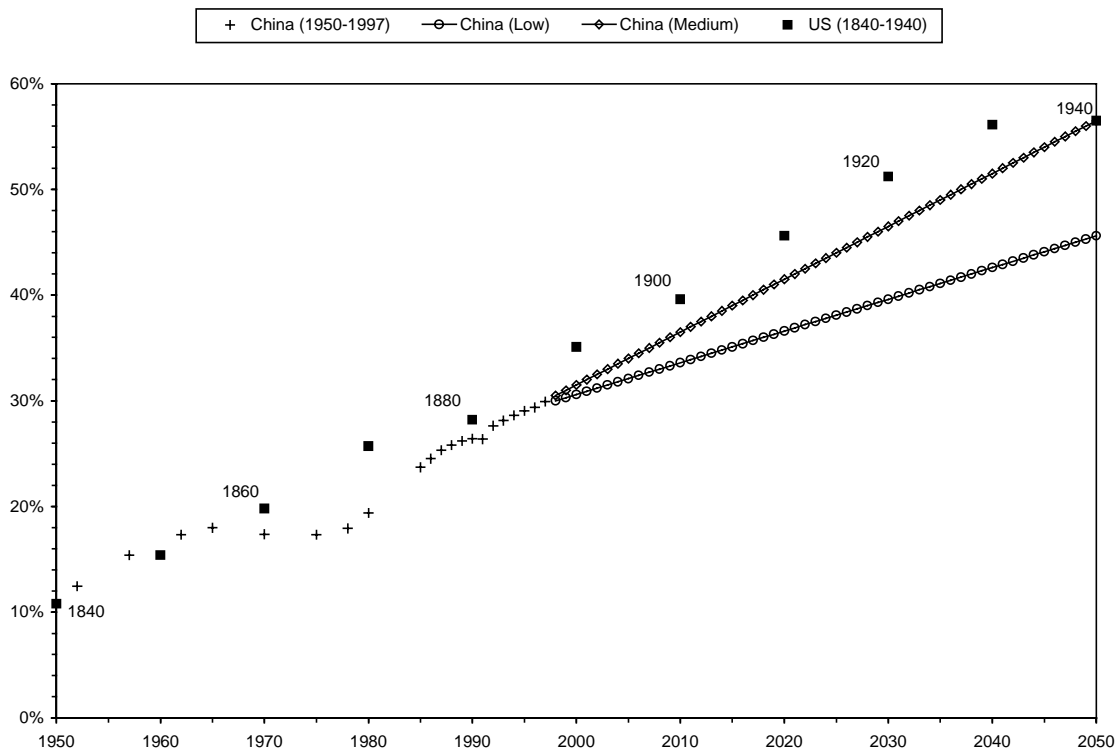
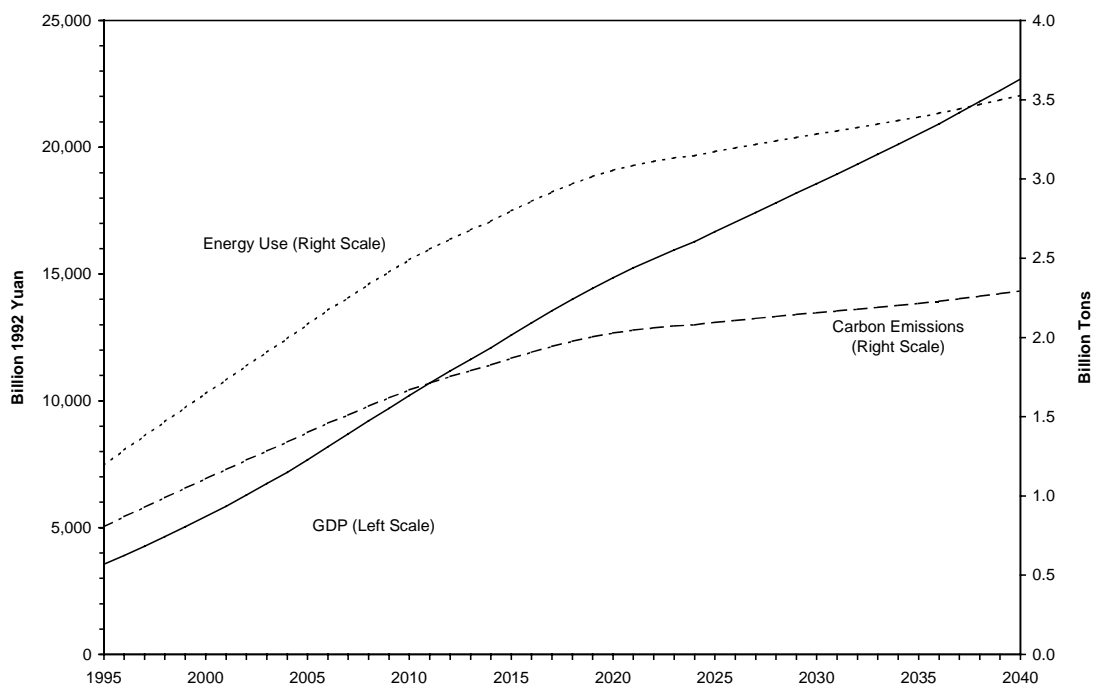


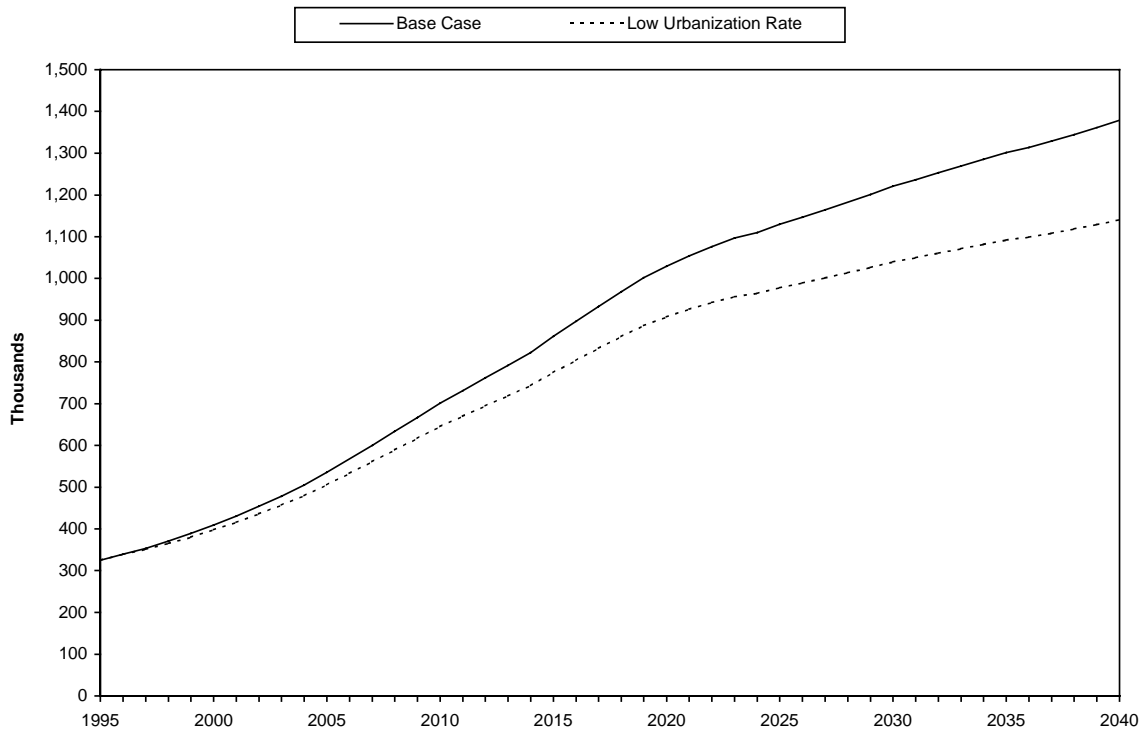
Figure A2. Projected GDP, energy and carbon emissions, 1995-2040



Note: Energy use is in standard coal equivalents (soe).

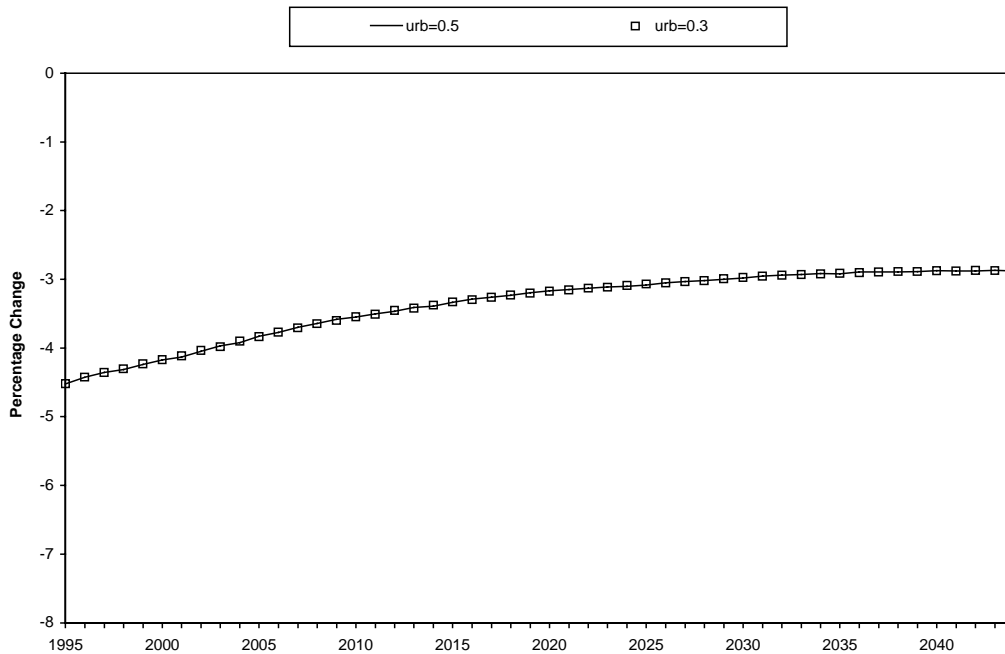


Figure A3. Excess deaths in base case versus low urbanization rate



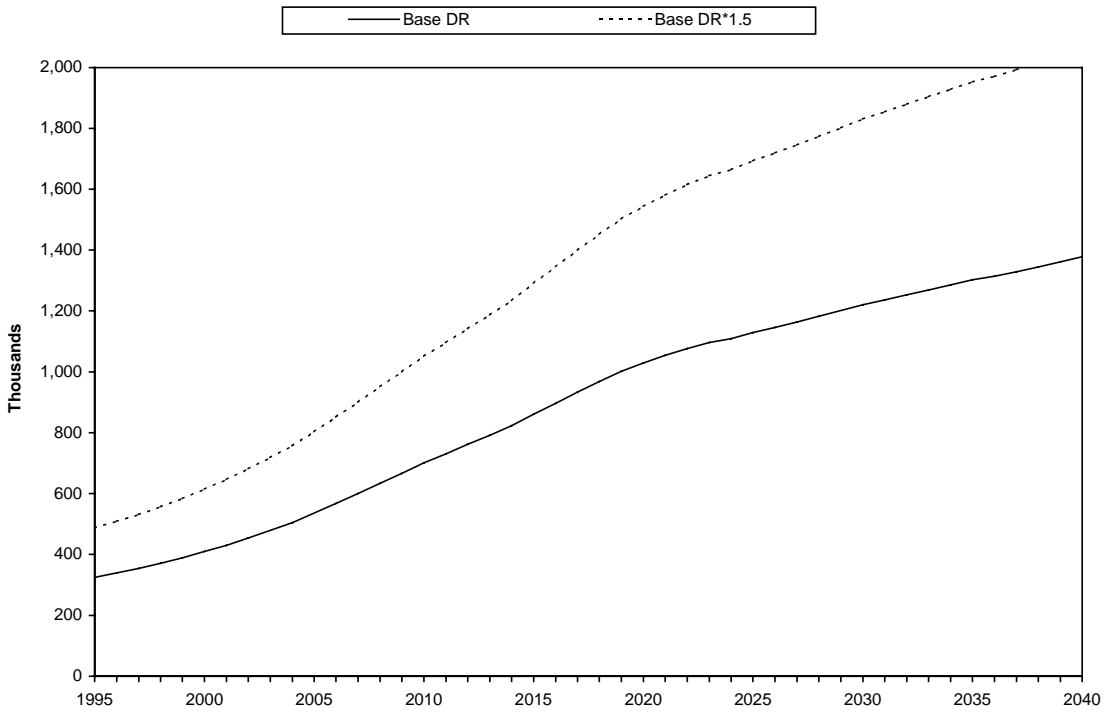
Note: The base case assumes that the urban share of the population is rising at 0.5% a year, in the low case we assume a 0.3% rate.

Figure A4. Change in excess deaths, base versus low urbanization case



Note: Each line represents the effect on premature deaths due to a 5% carbon reduction policy.

Figure A5. Excess deaths in base case versus high dose-response case



Note: The base case uses the central estimate of the dose response to a 1 microgram/m<sup>3</sup> increase in concentration. In the high DR case the value is 1.5 times the base coefficient.

# ANCILLARY BENEFITS ESTIMATION IN DEVELOPING COUNTRIES: A COMPARATIVE ASSESSMENT

by David O'CONNOR

## 1. Introduction

The possibility of reaping ancillary benefits from climate policy is generally accepted. The questions of their quantitative importance and how factoring them into the analysis might alter policy choices are still being actively explored. Thus far, most work has focused on public health benefits of reduced emissions of air pollutants associated with carbon dioxide (CO<sub>2</sub>) as by-products of fossil fuel combustion – SO<sub>2</sub>, NO<sub>x</sub>, suspended particulates, volatile organic compounds (VOC), carbon monoxide (CO), and ozone (O<sub>3</sub>). A smaller body of work has looked at other damages, notably crop damage from O<sub>3</sub>, forest damage from SO<sub>2</sub> and materials damage from that and sulphate aerosols. The early studies of ancillary benefits of reducing CO<sub>2</sub> emissions were done in Europe and the United States, where the prospect of quantitative restrictions on those emissions appeared to be imminent. More recently, work has begun to focus on developing countries. The policy rationale is similar, viz., that decisions about desirable levels of CO<sub>2</sub> abatement need to be informed by as full an accounting as possible of both costs and benefits, and that the nearer-term, more certain and local ancillary benefits may well carry more weight with policy makers than the longer-term, more uncertain and global ones from climate change mitigation.

This paper reports on work-in-progress at the OECD Development Centre (*DevCentre*) that makes use of computable general equilibrium (CGE) models to integrate ancillary benefits estimation into an economy-wide assessment of climate policy. To date, a study has been completed on the ancillary benefits of climate policy in Chile (Dessus and O'Connor 1999), and one is currently underway for India (Bussolo and O'Connor 2000 forthcoming). Both focus on public health benefits of climate policy in consequence of reduced local air pollution. A study on China is also planned, which will focus on the effects of reduced air pollution on crop yields, considering in the first instance O<sub>3</sub> but with possible extension in a second phase to include particulate haze. The paper compares, wherever possible, both the methodology and the results of this research effort with those of other studies undertaken for the same set of countries. In particular, Cifuentes *et al.* (1999) provides another set of estimates for Chile, using a "bottom-up" engineering approach, while Garbaccio *et al.* (2000) offers a CGE-based assessment of public health benefits of climate policy for China. These two studies provide useful comparators for the *DevCentre* studies, allowing the examination of the sensitivity of results to specific parameter or variable values. In conjunction with global studies like Abt Associates (1997), methodological analyses like Markandya (1998), and the European and U.S. empirical studies mentioned above, the developing country studies permit the exploration of certain hypotheses about cross-country and cross-regional differences in the relative magnitude of ancillary benefits.

The paper is organised as follows: Section 2 presents a simple analytical framework relating ancillary benefits to CO<sub>2</sub> abatement costs and pointing to the importance of different regulatory baselines in determining the magnitude of expected benefits. Section 3 takes a comparative look at methodological issues, notably in the context of the *DevCentre* studies and other CGE-based studies for developing countries. Section 4 compares results of various studies, discussing sources of significant variation, while Section 5 concludes with a reflection on what is needed to give policy makers greater confidence in the reliability and robustness of the estimates provided by this body of research.

## 2. Simple analytics of ancillary benefits

By climate policy we refer to any set of policies whose primary purpose is to slow the growth of net greenhouse gas (GHG) emissions (including through sink enhancement), including those that result in an actual reduction of such emissions relative to some base-year level (as for most Annex 1 Parties to the Kyoto Protocol). (In what follows, we focus on CO<sub>2</sub> abatement, as this is by far the most significant GHG in most countries.) Figure 1.A presents a stylised picture of how costs vary with the abatement level, suggesting that they increase at an increasing rate. In other words, *marginal abatement costs are increasing in abatement effort*. The figure also depicts a stylised ancillary benefits curve, which is shown as a ray from the origin with constant positive slope, suggesting as a first approximation that marginal ancillary benefits are equal to average ones. This follows from the epidemiological studies on mortality and morbidity effects of particulate exposure, many of which find that reductions in risk bear a roughly constant relationship to reductions in ambient concentration irrespective of the initial concentration level. The figure – and the subsequent analysis – abstracts from the primary benefits resulting from climate change mitigation, not because they are not considered important but because they are thought to be too uncertain and distant in time to influence significantly policy making in countries faced with more immediate and pressing concerns. Health of the population is one such concern, and while reducing air pollution exposure may not be the most urgently needed health intervention in countries where infectious diseases are rampant, in many parts of the developing world respiratory diseases are among the leading causes of mortality and morbidity – and air pollution is certainly among the aggravating factors in many cases. Indeed, acute lower respiratory infections rank first of all diseases in the world in terms of disability adjusted life years (DALYs) (WHO 1999), a measure which combines the burden from premature mortality with that from living with disability (Murray and Lopez 1996).

Through inversion of the net cost curve in Figure 1.A, Figure 1.B shows net benefits of CO<sub>2</sub> to be positive over some range, peaking at abatement rate  $a$  before declining, becoming zero at point  $b$  (the so-called “no regrets” rate of abatement) before turning steeply negative. An “optimal” climate policy would, needless to say, seek to maximise the net benefits (again bearing in mind the absence from consideration of primary climate benefits), and so “optimal” abatement would be somewhat lower than the “no regrets” rate.

The costs depicted in Figure 1 are those of limiting an economy’s emissions of CO<sub>2</sub>, which can be done only through one or more of the following: (a) reducing energy consumption; (b) fuel switching from high-carbon to low-carbon fuel; (c) lowering the carbon intensity of a given activity or set of activities; (d) reallocating resources away from energy- (specifically, carbon-) intensive activities. If the economy was operating efficiently in an initial equilibrium, any one of these actions involves an opportunity cost. It is only when one assumes pre-existing inefficiencies – e.g., in energy input per unit of output – that the gross abatement cost curve could be expected to dip below the  $x$ -axis over an initial range of abatement. In this event, the net cost curve also shifts down proportionally and the “no regrets” level of abatement is further increased.

The ancillary benefits curve is a construct involving several intermediate steps between the policy shock (say, a carbon tax) and the change in real disposable income, our welfare measure. These steps are depicted in Figure 2. The crucial link in the chain is from the carbon tax to the impact on other pollutants. Taking TSP for purposes of illustration, we need to know how a carbon tax – levied for example on the carbon content of fuel – translates into reductions in particulate emissions, in other words, the cross price elasticity of particulates with respect to carbon ( $\bullet_{pc}$ ). The higher is  $\bullet_{pc}$ , the greater will be the effect on particulate emissions of a given carbon tax. What determines the value of  $\bullet_{pc}$ ? Most importantly, it depends on the extent to which the two pollutants have been “de-linked” in the baseline through prior controls specifically targeted at particulates – in other words, on the stringency, and the strictness of enforcement, of particulate standards. Since growth in carbon emissions is still fairly closely linked to GDP growth (though with some variation in elasticities across countries), de-linking particulates emissions from carbon emissions implies de-linking their growth from GDP growth.

It is generally the case that the OECD countries (i.e., the bulk of Annex 1 countries under the 1997 Kyoto Protocol) have gone farther than developing countries in de-linking local pollution from GDP growth. Another way of putting this is that they have moved farther out along their inverted-U-shaped environmental Kuznets curves for pollutants like particulates and SO<sub>2</sub>. This observation suggests a hypothesis about the relationship between a given carbon tax and the size of expected ancillary benefits, viz., that *the lower a country's level of development, the larger are the expected ancillary benefits of a carbon tax*. This is because, given the limited prior abatement of local pollution, a tax on carbon translates into a bigger reduction in the more closely linked local pollutant. In short,

$$(GDP)_i < (GDP)_j \Rightarrow (\bullet_{pc})_i > (\bullet_{pc})_j \Rightarrow (AB_i | t_c) > (AB_j | t_c)$$

where  $t_c$  is the rate of carbon tax and  $AB_{ij}$  are the ancillary benefits for countries  $i$  and  $j$  (measured in physical units – e.g., premature deaths avoided per tonne carbon reduction). Whether this translates into larger monetised welfare gains depends on the relative incomes of the two countries, hence, on their respective willingness to pay (WTP) for the expected health improvements.

Figure 3 presents this analysis in graphical terms, showing the marginal abatement cost curves for local pollution for a low-income (MAC) and a high-income (MAC\*) country. The latter is shifted to the left because of the prior abatement of local pollution, so the response to a carbon tax already finds the high-income country on the steeply ascending portion of the MAC\* curve. Also in the high-income country, because of the relatively low cross price elasticity of carbon and local pollution, a given carbon tax translates into a lower effective tax on the latter –  $t_e^*$  versus  $t_e$ . The combination of these two effects implies a lower *post-tax* equilibrium level of local pollution abatement, hence, lower ancillary benefits in the high-income country than in the low-income one.

Figure 1A. **Gross and net costs of CO<sub>2</sub> abatement**

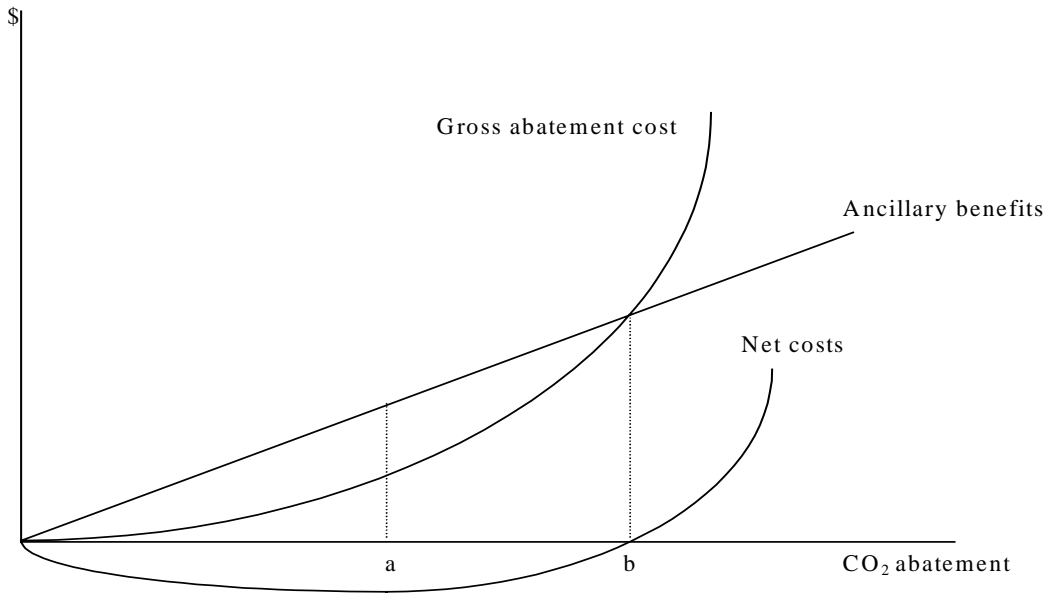


Figure 1B. **“Optimal” and “No Regrets” CO<sub>2</sub> Abatement**

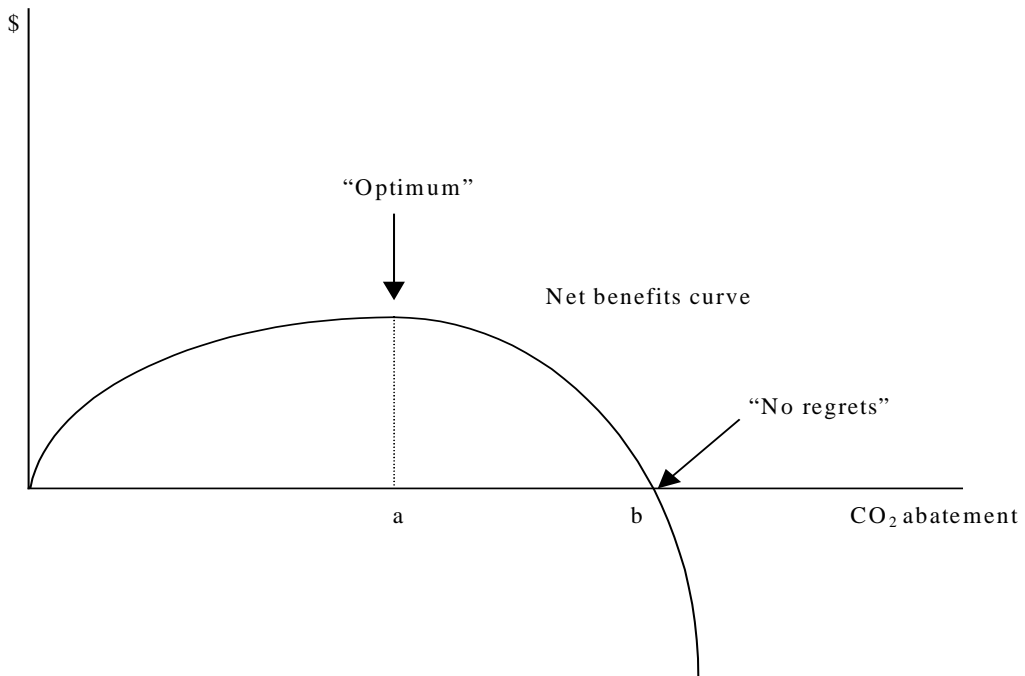


Figure 2. Links in chain from policy measure to welfare change

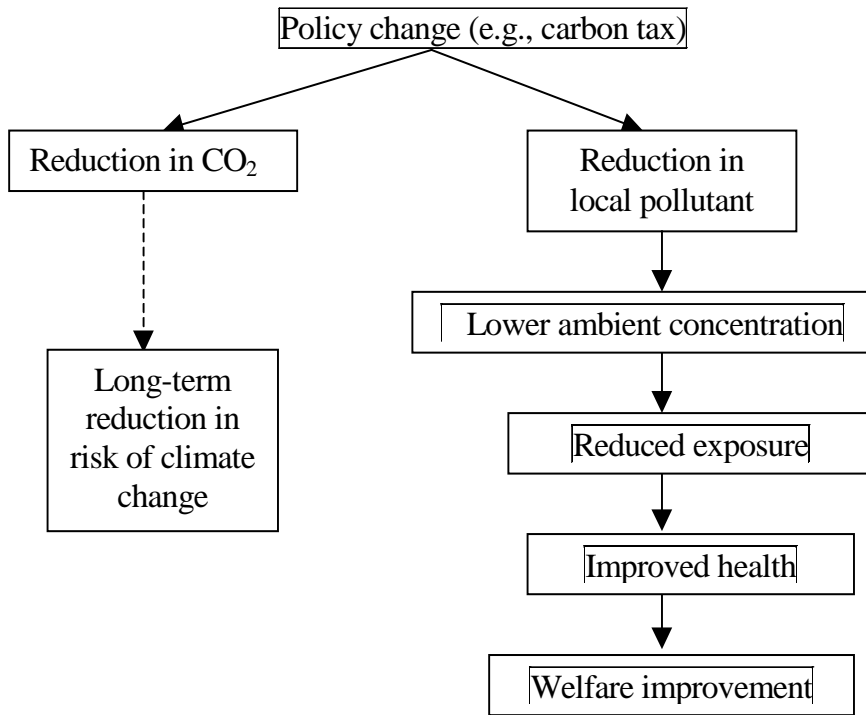
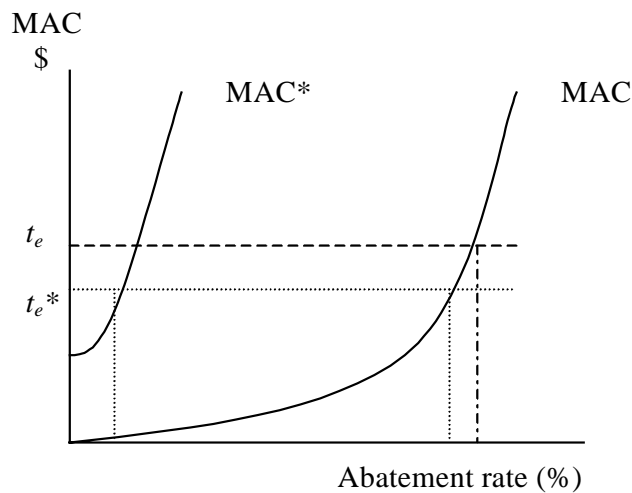


Figure 3. Marginal abatement costs and abatement rates for local pollutants, developed and developing countries



### 3. Methodological issues

Put simply, ancillary benefits analyses – like climate policy analyses more generally – can be dichotomised into top-down and bottom-up approaches. The *DevCentre* studies make use of the former, so that is the principal focus of discussion here, though at points reference is made to bottom-up methods for purposes of comparison. In any event, several of the methodological issues discussed relate equally to both types of approach.

The top-down approaches mostly make use of CGE models to look at economy-wide impacts of a given policy scenario relative to a no-policy baseline. These are the models of choice for most global climate policy modelling, where broad orders of magnitude of welfare change and rough comparisons across regions of the world are the most that is sought. Their principal virtue lies in their ability to capture feedbacks in the economic system, e.g., via relative price changes, that might lead to results other than those predicted from an examination of first-order, partial equilibrium effects alone. Their principal drawback is the paucity of technological detail, which makes them less than ideally suited for a thorough micro-level assessment of sectoral responses to a policy shock. Such models can, of course, be made more realistic, but at a cost in added model complexity. Another criticism levelled against CGE models is that they are not always strongly grounded in empirics – e.g., econometric estimation of the thousands of elasticities embedded in a typical model structure would simply be too data intensive, but on the other hand, elasticity values cannot simply be pulled out of thin air.

In the context of ancillary benefits estimation, one of the features of a typical CGE model is particularly noteworthy. It seldom incorporates a separate abatement technology for local pollutants. This implies that the only way to control those pollutants in the model is via inter-fuel substitution (e.g., switching from coal to gas in power generation) or via substitution of productive factors (e.g., labour) and/or other inputs for polluting energy in a given production process. At the level of the economy as a whole, structural change towards less polluting sectors can achieve the same results.

To the extent that end-of-stack or end-of-pipe abatement has already occurred, it is reflected in a reduced level of base-year emissions of local pollutants. In model simulations, however, further adoption of such technology cannot be readily accommodated. In reality, a carbon tax would most likely work its effects via fuel, factor and input substitution, so this is not a serious limitation in simulating climate policy. What is rendered difficult is any comparison of marginal costs of end-of-pipe/stack abatement with those of abatement via fuel/factor/input substitution. It is possible, for example, that in a country where no prior controls on particulate emissions are in place, there are capture technologies that would reduce emissions up to a point at significantly lower cost than would be possible incurred with a carbon tax. In that event, and referring back to Figure 3, model simulations would not be able to reflect movements along the shallow portion of the developing country's MAC curve but only along steeper sloping portions. In effect, MAC would be shifted leftward towards MAC\*.

Also – and it is here that the degree of technological detail matters – how well one can capture substitution possibilities – e.g., between fuels – depends on how disaggregated a model one has of the energy sector, hence, of fuel types. For instance, while most models distinguish coal and oil as separate sectors, few can distinguish low-sulphur diesel oil from high-sulphur diesel or low-ash coal from high-ash coal. From the perspective of carbon emissions, such distinctions are not particularly important, but they are when one is concerned with impacts of policy on local pollutants.



### 3.1 *Modelling emissions and dispersion*

Another difficulty with the use of CGE models for ancillary benefits analysis results from a lack of spatial detail. For global climate modelling, only the largest countries are treated individually, with the rest of the world grouped into broad regions. Ancillary benefits estimation for an individual country obviously requires a separate national CGE model, but even this is not necessarily adequate to capture the local-level dynamics that affect the size of ancillary benefits. Carbon dioxide is a global pollutant, so a single, undifferentiated national model suffices for analysing climate policy on its own, but the pollutants of interest for ancillary benefits estimation are mostly local or regional. Geographic location of emissions, stack heights of emitting sources, local temperature and meteorological conditions, population distribution and location of valuable assets vulnerable to pollution damage all matter to the nature and size of impacts<sup>90</sup>. Needless to say, this richness of detail is not well captured in a national model, especially for a large country. It is for this reason that, in the case of the *DevCentre* studies, it was decided that, while for Chile a single national CGE model would suffice (given that most air pollution impacts are concentrated in the heavily populated capital city of Santiago), for India a multi-region CGE model would be more appropriate.

In any event, it is seldom possible with a CGE model to achieve a degree of disaggregation ideal for analysing local air quality and health impacts, viz., at the level of the individual metropolis. Wedding CGE models to adequate air dispersion models remains a research challenge, not least because even a regional CGE does not usually incorporate a detailed locational grid of emissions within the region of the sort needed for more sophisticated air modelling. To illustrate the problem, suppose that, while coal-burning power plants account for 50 per cent of regional particulate emissions and motor vehicles 20 per cent, the latter contribute 60 per cent to ambient concentrations in the main regional metropolis, while the former contribute only 30 per cent. Ideally, this locational effect on the emissions-concentration relationship should be reflected in the basic dispersion model, but without the benefit of a source-receptor matrix, one might mistakenly conclude that a 10 per cent reduction in power plant emissions would reduce concentrations and exposure in the big city by 5 per cent.

Geographically localised ancillary benefits studies are able to incorporate more sophisticated dispersion models – e.g., of the Gaussian plume variety (see Colls 1997, ch.3, for a presentation of the Gaussian model with worked examples). This approach, which is rather data-intensive, is adopted in Cifuentes *et al.*(1999) for Santiago, Chile.

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<sup>90</sup> To illustrate the role of temperature and humidity, according to one estimate, concentrations of particulate matter from a fixed quantity of emissions in hot and dry regions are about one-third of what would be expected from the same emissions under most other climatic conditions (Working Group 1997).

### 3.2 Modelling concentrations and exposure

In the absence of detailed location-specific emissions data and a means of mapping these into ambient concentrations at different receptor points, some simplifying assumptions are needed to link emissions changes generated by policy simulations to changes in concentrations-exposures. In the extreme, for some pollutants (excluding those that are the product of atmospheric chemical reactions – e.g., O<sub>3</sub>, sulphate and nitrate aerosols) one can assume a simple linear relationship between regional emissions and ambient concentration measures in major cities, making use of base-year emissions and concentration data to determine the coefficient on emissions, and assuming some background level of emissions unattributable to specific sources. If information is available on the proportion of emissions-generating economic activities located in each metropolis, the regional emissions figures can be scaled down to approximate more closely local emissions. This is essentially the approach adopted in the Chile study of Dessus and O’Connor (1999). A slightly more sophisticated approach, used by Garbaccio *et al.* (2000) for China is to assign different coefficients to different sectoral groupings, depending on whether emissions from those sectors normally occur at or near ground level (e.g., motor vehicles and small boilers), from stacks of medium height (large industry), or from high stacks (power plants) (classification based on Lvovsky and Hughes 1998). In short, the dispersion function is of the form:

$$Conc_{TSP} = a + b_1 (Emis_{Tall}) + b_2 (Emis_{Medium}) + b_3 (Emis_{Low}),$$

where  $Conc_{TSP}$  refers to the average city-wide concentration of TSP,  $Emis_{Tall, Medium, Low}$  the region-wide or metropolitan-area-wide TSP emissions from each of three groups of sectors differentiated by typical stack height. The constant  $a$  is an approximation of the effect of background emissions on ambient air quality (in short, what concentration would obtain assuming zero sectoral emissions). The  $b_i$ s are the dispersion coefficients for emissions from each stack height, calculated using a simple dispersion model in which different atmospheric conditions are assumed to occur with given frequencies<sup>91</sup> and the key piece of additional data required is a metropolitan area’s radius (see Lvovsky and Hughes 1998). The use of even this somewhat more sophisticated dispersion model still involves a gross simplifying assumption, viz., that the specific geographic distribution of emission sources within the area does not significantly affect area-average pollutant concentration.

Even if location-specific emissions data are not available, it is clearly necessary to have an estimate of total emissions of a given pollutant and to be able to allocate those emissions by sector. Fuel-specific emission factors are a useful starting point. With sector-wise data on fuel consumption by type it should be possible to estimate maximum combustion-related sectoral emissions. To these one needs to add any process emissions not directly linked to fuel use – e.g., fugitive dust from cement plants. This total then needs to be adjusted downward by a factor reflecting removal with end-of-pipe/stack controls. Also, it is probably a reasonable assumption that, as enterprises invest in new plant and equipment, average emissions (per unit of fuel unit or per unit of output) will fall. Thus, in the case of particulates, and drawing on work by Lvovsky and Hughes (1997), Garbaccio *et al.* (2000) incorporate lower emission coefficients for new capital stock than for old, though no account is taken of the cost of engineering lower emissions into new capital equipment and emission coefficient reductions are treated as additional to the autonomous energy efficiency improvement (AEEI) factor.

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<sup>91</sup> Ideally, region- or city-specific information on atmospheric conditions can be found to determine these frequencies, but if not then certain “default” frequencies can be used as an approximation.

### 3.3 *Assessing health damages*

To assess actual health damages, ideally one would like to be able to measure effective human exposure – i.e., numbers of people exposed to what concentrations over what period. In practice, the epidemiological studies of pollution's health impacts mostly relate variations in ambient concentration to variations in relative risk, e.g., the risk of premature death. In short, if pollution (say,  $PM_{10}$ ) were reduced by a given amount from current levels, how many lives would be saved or, conversely, how many excess deaths would be caused by a given  $PM_{10}$  increase?

The results from multi-city U.S. studies of acute exposure to  $PM_{10}$  by Dockery, Pope and colleagues are quite consistent, finding an estimated 0.7-1.5 per cent increase in total mortality associated with a  $10 \mu\text{g}/\text{m}^3$  increase in  $PM_{10}$  concentration from mean levels in the range  $38\text{-}61 \mu\text{g}/\text{m}^3$  – i.e., several times lower than mean concentrations in many developing-country cities. A meta-analysis in Schwartz (1994) finds a consensus range for mortality increase estimates of between 0.7 and 1.0 per cent per  $10 \mu\text{g}/\text{m}^3$  increase in  $PM_{10}$  concentration. Comparing their estimates to those of other studies, Dockery *et al.* (1992) observe that the dose-response relationship between particulates and mortality is remarkably similar across a large range of concentrations, in a variety of communities, and with varying mixtures of pollutants and climatology. There is no evidence of a “no effects”, or threshold, concentration – at least not within the range observed in U.S. cities.

The robustness of the estimates is borne out by non-U.S. studies, including a handful in developing countries. For instance, Ostro *et al.* (1996) find a significant relationship, for Santiago, Chile, between ambient particulate concentration (in this case,  $PM_{10}$ ) and mortality, after controlling for confounding influences like temperature. In particular, the results from their basic OLS model suggest that a  $10 \mu\text{g}/\text{m}^3$  change in concentration around the mean ( $115 \mu\text{g}/\text{m}^3$ ) is associated with a 0.6 per cent change in mortality<sup>92</sup>. They note that their results are consistent with findings of various U.S. studies on the  $PM_{10}$  – mortality link and suggest that, for this reason, applying the U.S. estimates to developing countries may be appropriate where local research is not possible, assuming those countries are not drastically different from the United States in terms of variables like time spent outdoors, baseline health status, and medical care and access.

An aspect of the particulates–mortality relationship that can be important for impact valuation is the age distribution of those whose lives are foreshortened. In the U.S. studies, those at highest risk are the aged and infirm and also the very young. In India, by contrast, Cropper *et al.* (1997) find that, while the overall mortality risk is somewhat lower in Delhi than in the U.S. studies, the 15-44 age group are at greater risk than those over 65 years. For one thing, the proportion of the population in the latter age group is much lower than in the United States; for another, most of the deaths before that age are from causes unrelated to air pollution. The age profile of those at risk is clearly more pertinent when the measure to be valued is life-years-lost or DALYs than when it is premature deaths averted irrespective of remaining life expectancy.

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<sup>92</sup> The relationship between  $PM_{10}$  and mortality is non-linear, however, with a change evaluated at  $50 \mu\text{g}/\text{m}^3$  associated with a 1.4 per cent increase in mortality (i.e., closer to the U.S. means) and one evaluated at  $150 \mu\text{g}/\text{m}^3$  increasing mortality by 0.4 per cent.

Besides mortality, there are a variety of morbidity endpoints that may be affected by air pollution, though few relationships are borne out as consistently by the epidemiological literature as the  $PM_{10}$  - mortality link. There are two main ways in which the effects of pollution on morbidity are measured: as incidence of physical symptoms and illness and as behavioural responses to the symptoms/illness. The former are normally the object of interest in clinical studies, while epidemiological studies may report on symptom/disease incidence and/or effects on human activity. The most common measures for the latter are “restricted activity days” (RADs), “work loss days” (WLDs), hospital admissions, and emergency room visits. RADs are a more comprehensive measure than WLDs, including days spent in bed, days missed from work, and other days when normal activities are restricted due to illness (Cifuentes and Lave 1993). They are also a more subjective measure and thus subject to greater measurement error.

Reviewing briefly the epidemiological evidence (and drawing principally on Ostro, 1994), air pollution is most commonly associated with respiratory illnesses, though other illnesses linked to specific pollutants include cardiovascular illness and impaired neurophysiological development (in the case of blood lead in children). More specifically, particulate exposure has been found to be associated with lower-respiratory illness in children; particulate and ozone levels with exacerbation of asthma attacks among both children and adults; ozone in particular with eye irritation and respiratory symptoms; and long-term exposure to particulates and sulphate and nitrate aerosols with chronic bronchitis and reduced lung function.

### 3.4 Valuing impacts

There are three broad approaches to valuation of environmental benefits in general and ancillary benefits of climate policies in particular (see Freeman, 1993, for the classic text on valuation methods). The first approach tallies productivity losses or costs to the economy from illness, premature death, or damage to crops, materials and ecosystems. From a theoretical standpoint it is the least satisfactory, not being firmly grounded in welfare economics, i.e., on measurement of changes in individual welfare. It can, however, provide a lower bound estimate of “true” benefits. The other two approaches – the first based on revealed preferences and the second on stated preferences – avoid this problem. Of the two, more controversy surrounds the latter, since it is based not on observed but on hypothetical behaviour. This is not to suggest that revealed preference methods are problem-free; they are not (see following discussion).

In most valuation exercises, mortality benefits/costs tend to dominate morbidity benefits/costs, the main reason being the high value attached by most people to mortality risk reductions. These are captured by a concept known as the “value of a statistical life”, reflecting the willingness to pay of individuals for a given reduction in *ex ante* risk of premature death.

The literature purporting to estimate *VSL* is vast and still growing (see Viscusi 1993 for an earlier review). The overwhelming majority of the studies have been done in Europe and the United States. The bulk employ a revealed preference method called hedonics to estimate the compensating wage differential paid to those workers in jobs with relatively high fatality rates. From this one can derive an estimate of *VSL*. For instance, if it is found that, on average, a worker receives a wage differential of \$350 per year for assuming an added risk of accidental death on the job of 1/10,000, then this implies a *VSL* of \$3.5 million. When one transfers this estimate out of the context in which it was derived – e.g., to one of mortality risk from pollution – there are at least three possible sources of bias, two having to do with different risk characteristics and the third with different affected populations. First, assuming complete information, job-related risk is voluntarily assumed, while risk from pollution exposure is involuntary, in the nature of a negative externality imposed by others' behaviour (or a combination of own and others' behaviour). Second, the time dimension of the risks can differ. For example, certain risks from pollution exposure are delayed until later in life, and people may value differently risks avoided now to those avoided later. Trying to capture the notion of delayed risk in a contingent valuation (CV) questionnaire, Krupnick *et al.* (1999) find from pre-test results that the discounted *VSL* is significantly below estimates from hedonic wage studies.

Third and finally, reducing mortality risk from pollution may be valued differently by individuals according to their ages. The population sampled in hedonic wage studies consists of active workers, while those most adversely affected by air pollution (at least in the United States and other OECD countries) are beyond working age. The direction of any resultant bias is unclear, however. On the one hand, one might expect the elderly to be relatively risk-averse, while on the other the willingness to pay (WTP) to save relatively few extra years of life may be lower than WTP to save an average of 30 or more expected by active workers. One empirical study does find that the WTP of the elderly to reduce mortality risk is somewhat lower than that of younger persons (Jones-Lee *et al.*, 1985). Since, as noted above, Cropper *et al.* (1997) find for the Delhi population that those at greatest risk from particulate air pollution fall into the prime working-age group (15-44 years), using hedonic wage estimates of *VSL* may not be a significant source of age-bias, though the other sorts of bias mentioned above could still be present.

A separate issue crucial for ancillary benefits estimation in developing countries is the relationship between WTP for reduced mortality risk (or *VSL*) and *per capita* income. Naturally, the former can be expected to rise with the latter, but is the rise proportional? The reason it is important is that, for most developing countries, there are few if any on-site *VSL* studies comparable to the hedonic wage studies done for the United States. It is common practice, therefore, to borrow *VSL* estimates from the U.S. studies, scaling them for differences in *per capita* income between the United States and the target study site. What scaling factor should we use? If the ratio of *per capita* incomes is 10:1, and if U.S. *VSL* is \$3.5 million, should we assume a *VSL* in country *x* of \$350,000. Doing so implicitly assumes that the elasticity of *VSL* with respect to income is unity. This does not, however, square very well with the evidence. Rather, it would appear that *VSL* rises less than proportionately to income; in other words, its income elasticity is less than unity. In their benefits transfer study of air pollution in Central and Eastern Europe, Krupnick *et al.* (1996) assume an elasticity of 0.35 (based on contingent valuation studies reported in Mitchell and Carson 1986). Also, in their study of mortality risk valuation in India, Simon *et al.* (1999) find evidence that the *VSL* is higher relative to *per capita* income than for the United States, dismissing as implausible the possibility that Indian workers are more risk averse than their American counterparts. Also, studies of morbidity risk find a relatively low income elasticity of WTP to avoid illness, ranging from 0.26 to 0.60 (Loehman and De 1982 and Alberini *et al.* 1997).

In sum, when applying benefits transfer for either mortality or morbidity risk reductions, it seems reasonable to assume an income elasticity of WTP well below unity, indeed, probably closer to 0.5. Thus, the *VSL* in a low-income country will be lower than in a high-income one, but by less than the ratio of their *per capita* incomes would imply.

While strictly speaking, one should derive measures of WTP for both mortality and morbidity risk reductions from welfare-theoretic principles, in practice there are seldom subjective measures of WTP for all relevant health endpoints. In their absence, it may be necessary to rely on such observables as “cost of illness”. In any case, once mortality benefits and morbidity benefits of a policy change have been calculated, they need to be incorporated back into the economy-wide model to determine the resulting welfare change. In practice, this is done by calculating what reduction in disposable income would leave individuals indifferent between the *status quo* and a post-policy state with reduced air pollution, fewer premature deaths, and improved health. That change in disposal income represents the amount of the welfare gain from cleaner air, and it can be compared in turn to the costs of achieving that improvement, measured by the reduction in disposable income associated with a carbon tax or other policy (and abstracting from any ancillary benefits).

#### **4. How far do results differ across studies and why?**

The literature on ancillary benefits of climate policy dates at least to the early 1990s (cf. Ayres and Walter 1991). Ekins (1995, 1996) reviews a number of ancillary benefits studies, with an emphasis on Europe. While there are relatively few, they yield widely varying estimates. In part, the variance stems from different underlying estimates – e.g., of *VSL* – but there are additional sources: differences in the method of estimating benefits (e.g., damages avoided *versus* abatement costs avoided); differences in scope of benefits included (e.g., some studies include both emission-related benefits and non-emission-related ones like the reduction in traffic congestion, accidents, and noise resulting from reduced road transport); differences across study sites in population exposure to pollution (with generally higher population densities in Europe than in the United States and prevailing winds blowing pollution inland in Europe, but out to sea from the eastern United States); the assumed stringency of CO<sub>2</sub> control; the timeframe of the scenario and the date of measurement of benefits. Another potential source of variation are differences in definitions of local pollutant baselines, deemed to be important in U.S. studies reviewed by Burtraw and Toman (1997; see below). Considering only emission-related benefits, the values range (in 1990 \$US) from a low of \$20/tC (Barker, 1993, for the U.K., based on social preferences revealed from the marginal costs of implementation of existing abatement technologies) to a high of \$212/tC (Alfsen *et al.* 1992, for Norway; Pearce 1992 reports a similar figure for the U.K.: \$195/tC). A mean value of emission-related benefits, based on the estimates reported in Ekins (1995), is around \$100/tC.

With the exception of Ayres and Walter (1991), the other studies reviewed in Ekins (1995) were conducted in Europe. Ayres and Walter find ancillary benefits for the United States of around \$23/tC (at 1990 prices) from fossil fuel emissions reductions in two sectors<sup>93</sup> (transport and electricity, which together account for about two-thirds of carbon emissions). While Ekins discounts this study because of its partial coverage, a more recent review for the U.S.A. by Burtraw and Toman (1997) reports on the results of eight studies whose mean estimate of ancillary benefits is virtually identical to the Ayres and Walter figure (i.e., \$24/tC), with a low estimate of \$2.64 and a high of \$78.85. Burtraw and Toman state a preference – on methodological grounds – for estimates at the lower end of the range (i.e., below \$7/tC) (though these cover only the electricity sector; the two economy-wide model-based estimates are above the mean but neither yields the highest estimate). Those studies producing the lower estimates also assume weaker control measures and smaller carbon reductions. Estimated abatement costs per tonne carbon reported in these studies are in the \$10-20/tC range, so with ancillary benefits in the \$3-\$7/tC range (and with a similar differential between ancillary benefits and control costs found in studies assuming somewhat more vigorous abatement), Burtraw and Toman propose a “rule-of-thumb” (for the United States) that ancillary benefits can be assumed to be roughly 30% of the cost of carbon reduction for low to moderate rates of abatement. Observing that over some range the marginal costs of GHG reductions are likely to be close to zero, the authors conclude that the existence of ancillary benefits even as small as \$3/tC could significantly increase the volume of emissions reduction that is considered “no regrets” in the sense of having negative or zero net cost.

Starting with the less optimistic assumption (from Nordhaus 1991) that even at low abatement levels costs are positive, Ekins (1995) calculates marginal abatement costs for small increments in the abatement rate, using an equation estimated by Nordhaus in a regression of the marginal abatement costs from several studies on their respective abatement rates. In so doing, he finds that even the lowest estimate of ancillary benefits on which he reports (\$23/tC) would justify a CO<sub>2</sub> emissions reduction of nearly 15%, supporting the Burtraw and Toman conclusion.

There does not yet exist the same wealth of studies on developing countries, though a few have been completed and several more are in progress (the two major endeavours being spearheaded, respectively, by OECD Development Centre and the U.S. Environmental Protection Agency through its *ICAP*<sup>94</sup> initiative). Thus far, each has produced an ancillary benefits study for Chile (Dessus and O’Connor, 1999, and Cifuentes *et al.*, 1999). The former is presently undertaking a study of India (Bussolo and O’Connor 2000 forthcoming) and will soon initiate one for China, while the latter has several in progress – for Argentina, Mexico, China and Korea. Garbaccio *et al.* (2000) have produced preliminary results for China and Joh (1999) for Korea. A preliminary comparison of results of these studies – with each other and with the OECD-country studies – suggests that there is also a fairly wide range in value estimates of ancillary benefits, even for the same country (e.g., Chile), but that estimates of physical benefits (e.g., numbers of deaths averted) are less variable.

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<sup>93</sup> The benefits estimates are based on the assumption of a 20% reduction in air pollution from 1978 levels.

<sup>94</sup> International Co-Control Analysis Program, initiated in 1998, benefits from financial and technical support provided by The National Renewable Energy Laboratory (NREL) and the World Resources Institute (WRI), as well as other contractors such as Abt Associates, all of the United States.

The latter is understandable in view of the fact that most ancillary benefits studies draw upon the same set of dose-response functions gleaned from the epidemiological literature reviewed briefly above. Thus, their assumptions about the effects on mortality or morbidity of a given reduction in ambient concentration of local pollutant  $x$  are very similar. For ancillary benefits analysis, as described in the preceding section, it is necessary to work back from changes in concentration (say, of particulates) to changes in particulate emissions to changes in carbon emissions. Only then can we calculate the ancillary benefits (in physical terms) per tC reduction. Table 1 reports on calculations of this measure based on several ancillary benefits studies. The first two are for Chile, and they show a number of lives saved (or premature deaths averted) ranging from 89 to 100 per MtC reduction. Bearing in mind the discussion of Figure 3 above, it is interesting to compare the results for Chile with those for the United States (with a significantly higher *per capita* income and stricter baseline environmental standards) and for China (with lower income and more lenient standards). The ordering of the size of mortality benefits – China highest, U.S.A. lowest, and Chile in between – is consistent with our hypothesis that the ancillary benefits (measured in physical units) of climate policy are likely to be higher the lower a country's baseline air quality.

Translating physical benefits into monetary values requires, in the case of mortality benefits, an estimated *VSL*. Since this is closely related to *per capita* income, it is naturally lower in poorer countries than in richer ones. Thus, if physical benefits are larger in the former than the latter, but *VSLs* are lower, there can be no *a priori* expectation about whether the monetary value of ancillary benefits per tC reduction will be higher or lower. If differences in physical benefits are sufficiently large, or differences in *per capita* income sufficiently small, the larger physical benefits in the poorer country could translate into larger monetary benefits than in the richer one (or at least comparable ones). The *per capita* incomes are likely to be closer in terms of purchasing power parity, or *PPP*, than in terms of market exchange rate conversion (in some cases significantly so). Thus, the choice of conversion rate to a common currency can make an important difference to the results. In our view, *PPP* is the more appropriate rate since the trade-off of mortality risk against other items in an individual's utility function depends on real disposable income, which in turn depends on *real* purchasing power of goods and services.

Table 2 presents three different sets of *VSL* estimates for Chile, with the underlying assumptions. It illustrates well the importance of two choices: of the "best estimate" of *VSL* from U.S. studies for purposes of benefits transfer, and of the exchange rate to be used in converting *VSLs* to a common currency. Dessus and O'Connor (1999) employ a moderate *VSL* estimate from U.S. studies, while Markandya's (1998) figure of \$4.8 million is at the high end of the range and Cifuentes *et al.* use a conservative estimate (\$1.9 million). Dessus and O'Connor (1999) and Markandya (1998) both employ a *PPP* exchange rate, while Cifuentes *et al.* (1999) employ the 1995 market exchange rate. The combination of the high U.S. *VSL* and the *PPP* exchange rate make the Markandya estimate for *VSL* in Chile (1995) more than three times higher than Cifuentes' estimate. In a pairwise Dessus/O'Connor–Cifuentes *et al.* comparison, the discrepancy in 2010 *VSL* estimates for Chile arises essentially from one source: the different exchange rate, or conversion factor, used, with Cifuentes *et al.* using a ratio of U.S. to Chilean *per capita* income (1995) of 5:1, based on the market exchange rate, and Dessus and O'Connor using a ratio of 2.5:1 based on a *PPP* exchange rate taken from the World Bank's 1999 World Development Indicators (CD-ROM). In the 2000 version of WDI, Chile's 1995 *PPP per capita* income has been revised downward substantially, so the new ratio of U.S. to Chilean *PPP* income is 3.8, i.e., much closer to the ratio used by Cifuentes *et al.*<sup>95</sup>.

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<sup>95</sup>

Moral: if you want to do benefits transfer right, do your own estimation of *PPP* GNP!



Ideally, then, one would not need to rely on benefits transfer for estimating the *VSL* for use in a particular ancillary benefits study. As we have just seen, this may pose a problem if there is some uncertainty about estimated *PPP per capita* income. In addition, there remains some uncertainty about the income elasticity of *VSL* to apply in making the transfer. In Table 2, the assumed elasticity is unity, while as discussed above there may be reason to suppose it is much lower. Cifuentes *et al.* are apparently attempting their own contingent valuation survey to estimate WTP for reduced mortality and morbidity risks in Chile. Likewise, the India ancillary benefits study currently in progress (Bussolo and O'Connor 2000, forthcoming) can avail of the results of the Simon *et al.* (1999) hedonic wage study for Delhi which, as noted above, yields a *VSL* strongly at odds with a unitary income elasticity assumption.

In summary, a first best strategy for ancillary benefits valuation is to rely on original WTP estimates for the study site. Benefits transfer is distinctly second-best and fraught with potentially significant biases.

Table 1. **Comparison of mortality benefits estimates of CO<sub>2</sub> reductions**

<i>Study</i>	<i>Lives saved per MtC reduction</i>	<i>Scenario Assumptions</i>
Cifuentes <i>et al.</i> (1999)	89	Chile, 2020: 13% CO <sub>2</sub> reduction
Dessus and O'Connor (1999)	100	Chile, 2010: 10% CO <sub>2</sub> reduction
Garbaccio <i>et al.</i> (2000)	430	Chile, 2010: 15% CO <sub>2</sub> reduction
Abt Associates (1997)	82	USA, 2010: 15% CO <sub>2</sub> reduction

Table 2. **Comparisons of value of a statistical life (VSL) for Chile**

<i>Dessus/O'Connor</i>	<i>Markandya</i>	<i>Cifuentes et al.</i>
\$2.1 million (2010)	\$1.37 million (1995); \$2.1 million (2010)	\$ 375,000 (1995); \$ 780,000 (2010)
1992 PPP \$	1994 PPP \$	1995 \$ at market exchange rate
Ratio to 1995 U.S. VSL (\$ 2.6 million) = 0.79	Ratio to U.S. VSL (\$ 4 million) = 0.34; 0.53	Ratio to U.S. VSL (\$ 1.9 million) = 0.2; 0.41

Source: Dessus and O'Connor (1999), Markandya (1998), Cifuentes *et al.* (1999).

Notes: World Bank estimate of 1995 ratio (Chile/U.S.A.) of per capita GNP = 0.4.  
Assumed elasticity of VSL with respect to income (GNP) = 1 in all cases.

## 5. How to enhance policy relevance

The preceding discussion leaves some doubt about how useful the results of ancillary benefits estimation may be to policy makers, given the wide variation observed. A few general observations are offered here on the issue of credibility and how to enhance it (see also Davis, Krupnick, and Thurston 2000 for a discussion of the credibility of ancillary health benefit and cost estimates). Pearce (2000) notes that, in OECD countries, the ancillary benefits studies conducted so far have been largely academic and poorly integrated at best into climate policy making. The same could be said, but perhaps even more so, in developing countries.

To be credible, any study of ancillary benefits must make its assumptions, methodology and scope as transparent as possible. The most likely criticism of results is that the benefits of climate policy are exaggerated by unrealistically optimistic assumptions, by neglect of certain costs, by an inappropriate baseline definition, or by incorrect estimation procedures. With respect to the baseline, it is perhaps best to give government the benefit of the doubt, i.e., assume that it will be successful in implementing any significant local environmental regulations already on the books or soon to be introduced. A useful method of dealing with parameter uncertainty is to conduct sensitivity analysis, varying parameter estimates over some range. To the extent that there are estimates from multiple studies, one can perhaps assume a sampling distribution to obtain a confidence interval around the mean and calculate benefits using parameter values marking the extremes. Clearly, the greater the variance in parameter estimates across studies, the wider the confidence interval will be.

In presenting results of an ancillary-benefits study, it is helpful to bear in mind the policy use to which it is to be put. The principal utility of such a study is to provide order-of-magnitude estimates of how large the expected local health (and/or other) benefits are from a given climate policy. This information may be valuable in and of itself and, in addition, it may be useful to be able to compare the value of ancillary benefits with the costs of GHG abatement (also on occasion with the primary benefits of climate change mitigation.) While for cost-benefit comparisons valuation of ancillary benefits is essential, in general, it is strongly recommended to report the physical impact estimates as well (premature deaths avoided, DALYs, symptom-days avoided, etc.), since these are apt to be somewhat less uncertain than the monetary values.

Where ancillary benefits estimation is done in a developing country, it may not be possible to find local epidemiological studies and/or WTP studies on which to base the analysis. In this case, one needs to proceed with extreme caution in undertaking benefits transfer. Transferring parameter estimates from epidemiological studies done elsewhere seems to be warranted (at least in the case of particulates), but even here there is the possibility that the age distributions of those affected may differ between original study site and new study site (recall the case of particulate-related mortality in Delhi *versus* U.S. cities). Potential problems with benefits transfer are more serious with respect to *VSL* and WTP for morbidity risk reductions. They include, to recap: (i) obtaining reliable estimates of *PPP per capita* income; (ii) choosing a suitable income elasticity of *VSL* for converting the chosen foreign *VSL* to a local one; (iii) ensuring reasonable comparability of populations and of risks between the original *VSL* study context and the pollution context of the ancillary benefits study. Ultimately, if resources permit, there is no good substitute for generating original WTP estimates at the new study site. If not, one may want to consider, in addition to *VSL* transfer from elsewhere, calculating from local earnings data the foregone earnings from premature mortality, as a lower bound on mortality benefits (bearing in mind the well-known limitations of this “human capital” measure).

To provide a rough check on the plausibility of estimates obtained, it is useful to compare results with those of other studies in terms of a common metric. This applies both to the underlying *VSL* and *WTP* estimates employed in the study and to the final value of ancillary benefits. In the first case, a common practice is to compare *VSL* to *per capita* income of the country; another is to compare it to the present discounted value of foregone earnings as a result of premature death (in effect, comparing the age-adjusted *VSL* from hedonic wage or contingent valuation studies – or benefits transfer based on such studies – with the human capital measure). Where comparable ratios are available for several countries, this provides a broad consistency check, though an interpretation is not easy in the event that the ratios show no tendency to converge.

In the case of ancillary benefits *per se*, the most common metric is the value of benefits per tC reduction. As previously noted, this measure tends to vary fairly widely across regions, though within regions – e.g., among the U.S. studies alone – the variation is less extreme. There can be significant factors explaining inter-regional differences (population density, climatology, *per capita* incomes, etc.), but wide divergence of results within a given region or country is apt to be more problematic, requiring careful diagnosis of the source of discrepancy.

Finally, an important element of ancillary benefits analysis in developing countries should be the consideration of the relative merits (in particular relative costs) of climate policy *versus* more direct measures to control local pollution. As explained above, this is especially pertinent in the developing-country context because one cannot necessarily assume that low-cost abatement options, e.g., for particulates, have already been exhausted. In short, one would like to be reasonably sure, when pointing to the ancillary benefits of climate policy, that there are not significantly lower-cost options for a developing country to realise those same benefits.

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# REDUCED DAMAGE TO HEALTH AND ENVIRONMENT FROM ENERGY SAVING IN HUNGARY

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## 1. Summary

The aim of this study was to assess the costs and benefits of the implementation of a specific energy saving program in Hungary. The program constituted the major part of measures to meet Hungary's obligations under the UN Framework Convention on Climate Change, and was primarily designed to reduce the emissions of CO<sub>2</sub>. The energy saving expected from the program was approximately 64 PJ/year (7.7% of the total energy consumption).

We have estimated the possible reduced damage to public health, building materials, and agricultural crops that may be obtained from implementing the program. Possible benefits were estimated using a bottom-up methodology where we applied monitoring data and population/recipient data from Hungary and exposure-response functions and valuation estimates (unit prices in terms of damage costs and willingness-to-pay) mainly from US and Western European studies. Our analysis indicates that the main benefit from implementing the program relates to public health, and that reduced prevalence of chronic respiratory diseases and premature mortality are the most important effects. The estimated annual benefit of improved health conditions alone, as estimated by a bottom-up willingness-to-pay approach, is approximately 650 mill. US\$ (370 mill. US\$ - 1170 mill. US\$) and is likely to exceed the investments needed to implement the program. In addition there are significant benefits due to reduced replacement and maintenance costs for building materials (30 - 35 mill. US\$ annually in Budapest only). Crop damage from ozone may be large, but a significant improvement in Hungary depends upon concerted actions in several countries.

A marginal benefit of reducing CO<sub>2</sub> emissions has been suggested by others to be in the range of approximately US\$ 64 to US\$ 164 per ton carbon. Applying these estimates to the Hungarian case study gives an annual benefit from reduction in CO<sub>2</sub> emissions with respect to climate change in the range 86 - 222 million US\$, i.e. lower than the ancillary benefits.

Information from the bottom-up assessment was used in a macroeconomic model to carry out an evaluation of the social costs of the energy saving program. The optimal level of abatement was estimated to be about 65 PJ/year, i.e. slightly higher than the original program. A leakage of about 5 PJ was estimated to occur due to improved environment, hence the total energy saving is estimated to be around 60 PJ/year. The leakage is due to increased output in the commodity and service sectors following lower damage, which leads to increased energy use. The value of the Energy Program in the macroeconomic study was estimated at 2.6 cents/kWh, which compares with 3.6 cents/kWh in the bottom-up study based on the willingness to pay. The reason for the discrepancy between this estimate and the bottom-up estimates is that the unit prices applied in bottom-up studies do not reflect the change in marginal cost when the environment improves. This is because the valuation is usually carried out prior to the implementation of the measures, i.e. in a situation when the abatement level is sub-optimal. Since such unit prices are based on constant prices, and the marginal cost of improving the environment is likely to be reduced as measures are implemented, the result may be that the total value of abatement is overestimated in bottom-up studies.

## 2. Introduction

In environmental policy making there is a continuous need to evaluate alternative measures to improve the quality of the environment. In doing so, the demand for reliable economic assessments increases. To provide reliable assessments, there ought to be consensus about the methodology of valuation. This is the point at which the controversies among economists start, and by use of different methodologies one may end up with widely different estimates of benefits for the same environmental improvement. In a case study in Hungary different approaches were taken to estimate the benefits that may be achieved by improving the environment.

First, a bottom-up study (impact pathway) was made, which included estimated benefits in terms of reduced damage to health, building materials and cereal crops. The physical estimates were evaluated by means of willingness-to-pay and damage cost unit prices, respectively. Secondly, a model for integrating damage costs and demand side assessments from bottom-up studies into a macroeconomic framework was developed, in order to 1) estimate the optimal level of abatement; and 2) to estimate the value of environmental improvements. The present paper is mainly based on Aunan *et al.* (1998) which describes the bottom-up assessment and Aaheim *et al.* (2000), which describes the macroeconomic assessment.

The focus of our study was the benefits that may be achieved from implementing measures that reduce the overall energy consumption, and thereby the general pollution level, in Hungary. The measures are described in the National Energy Efficiency Improvement and Energy Conservation Program (NEEIECP). The concept of the program was elaborated by the Ministry of Industry and Trade and accepted by the government in April, 1994. The program constitutes the major part of measures to meet Hungary's obligations under the Framework Convention on Climate Change (Poós, 1994; Pálvolgyi and Faragó, 1994; OECD/IEA, 1995). Very briefly the main goals are to:

- improve environmental protection;
- reduce the dependency on imports;
- save domestic energy resources;



- postpone construction and installation of new base load power plants;
- increase the competitiveness of the economy;
- adjust to the energy policy of EU and to the OECD/IEA recommendations.

Our scenario was assessed over 5 years and had the baseline economic assumptions that the annual growth rate of GDP was expected to decrease up to 1995. Beyond 1995 the annual growth rate was assumed to increase to 1-2 % per year (in 1995 the growth rate was ca. 1,4% and in 1996 it was 0,8-0,9%). Moreover, it was assumed that 1) the price system of energy carriers should reflect realistic expenditure and the cross financing should be ceased; 2) energy awareness should be developed as a consequence of rise in prices of the energy carriers; and 3) Centralised subsidy and international aid programs (e.g. PHARE) should be assisted through a soft loan system. The following estimates were given for the scenario:

- Saved energy: 63.7 PJ/year.
- Saved energy cost: 373 mill. US\$/year.

Whereas the saved energy (in terms of PJ) was allocated to the various sectors and measures, the estimated total investment (capital and operating costs) needed during the implementation period of 5 years was given as an aggregate, the present value being 422 mill. US\$ (i.e. the program seems to be highly profitable, as the present value of the saved energy cost would be above 5 bill. US\$ if we assume a lifetime of the measures of 15 years). We had some information on the estimated relative needs within some sectors, but it proved difficult to obtain a comprehensive picture of the cost estimate. This weakness of the program, as well as large uncertainties in the estimated energy saving potential, have also been pointed at by OECD/IEA (1995).

Table 1. **Reductions in annual energy consumption and emissions, estimated to result from implementation of the NEEIECP, relative to 1992**

	<b>Reduction</b>	<b>% of total</b>
Energy use (PJ)	63.7	7.7
TSP (ktons)	10.1	9.3
SO <sub>2</sub> (ktons)	46.8	5.7
N <sub>2</sub> O (ktons)	0.5	7.8
CH <sub>4</sub> (ktons)	1.1	9.4
nmVOC (ktons)	5.8	10.0
CO (ktons)	71.8	12.3
NO <sub>x</sub> (ktons)	17.4	10.1
CO <sub>2</sub> (ktons)	3800 - 4920*	5.8 - 7.5

The possible energy saving and estimated emissions reductions are given in Table 1. The emission coefficients for each sector are from Tajthy and co-workers (Tajthy, 1993; Tajthy *et al.*, 1990). In the study it was assumed that the energy saving primarily affects the consumption of fossil fuels, and we also assumed a status quo baseline emission scenario (see Aunan *et al.*, 1998 for a discussion). Another important assumption is that a given per cent overall reduction in the emissions of an air pollutant gives the same reduction in average concentration level in the cities. Although this assumption may be a reasonably good approximation in view of the aggregated level of the calculations done in the study, it would not be valid for large reductions. In those cases contributions from the regional background concentration level (also caused by transboundary pollution) should be considered. To illustrate the approximate share of the concentration level of air pollutants that may be inert to national emission reductions, we used data on the share of deposition of nitrogen and sulfur that are indigenous (EMEP, 1996). Using Budapest as an example, we estimated a likely range for the background level of PM<sub>10</sub> and a likely range of the contribution to this level from foreign sources. The calculations indicated that around 62% (49%-67%) of the annual average concentration level of PM<sub>10</sub> in the city would be influenced by national emission reductions. A recent study by Tarrasón and Tsyro (1998) showed that the relative indigenous contribution generally is smaller for PM<sub>2.5</sub> (the fine fraction of airborne particles) than for sulfur and nitrogen deposition.

### **3. Reduced health effects as estimated in the bottom-up study**

#### **3.1 *Exposure-response functions from epidemiological studies***

For an individual the exposure to air pollution may vary considerably over time. The indoor and outdoor micro-environment concentration levels vary according to e.g. the pollutant sources and dispersion patterns. A person's level of activity is among the factors determining the dose that enters the body. Additionally, the susceptibility varies among people, according to for instance age and health status. Hence, the risk of adverse health effects from air pollution is by no means equally distributed in a population.

Although some of the exposure-response functions for health effects and air pollution used in this study apply to specific groups, as elderly or children, they in most cases only provide estimates of average frequencies of health effects on a population basis. For instance there is no distinction between four persons having a one day illness episode and one person having a 4-day episode. It should be kept in mind that a subgroup of more susceptible individuals suffers a disproportionate share of the damage.

Epidemiological studies provide the best basis for establishing exposure-response functions for health damage in a population due to air pollution, because they generally apply to a cross-section of the population regarding age, gender, sensitive sub-populations, and also regarding the personal exposure level relative to the average pollution level. The exposure-response functions used here employ one indicator component for each effect type, and are mainly based on a review of epidemiological studies primarily from Western European countries and USA (Aunan, 1996). There are several problems connected to transferring risk estimates from one population to another (see Aunan *et al.*, 1998).

Many exposure-response functions for health effects of air pollution relate to the concentration of suspended particles. Because particles are monitored only in a limited number of Hungarian cities (representing, however, 57% of the urban population), we investigated whether we could obtain reasonable estimates of the particle level from data on other pollutants which were available for more cities (representing 83% of the urban population). We found that the NO<sub>2</sub>-data could be used. In Budapest NO<sub>2</sub>, PM<sub>10</sub>, and TSP are monitored, in the 18 county capitals NO<sub>2</sub> and TSP are monitored, whereas in the remaining cities only NO<sub>2</sub> is monitored (in addition to SO<sub>2</sub> and dust fallout, which are monitored in all cities). We used the data from Budapest to establish the relations for NO<sub>2</sub> versus PM<sub>10</sub>, and PM<sub>10</sub> versus TSP, and used the data from the county capitals to test these relations (see Aunan *et al.*, 1997). We found a PM<sub>10</sub> /TSP-ratio of around 0.35-0.4, which is lower than what is often reported in studies in USA, where 0.5 - 0.6 is suggested as a conversion factor if no other data are given. Other studies have also indicated that the PM<sub>10</sub>/TSP-ratio is lower in CEE than in Western Europe and the US (Clench-Aas, and M. Krzyzanowski, 1996). We concluded that our method, by estimating the concentration of particles from the NO<sub>2</sub>-level, probably at the most underestimates the response by 14% (Aunan *et al.*, 1997). We also concluded that, although the concentration of particles may be underestimated in cities with high levels of particles, this is less serious when we have in mind the purpose of the approximation procedure, which is to be able to assess possible benefits from energy saving measures. In cities with very high TSP levels process emissions from industry are an important source, and probably these emissions are less influenced by pure energy saving measures. The particle concentration estimated from the NO<sub>2</sub>-data may simply be regarded as the level caused by combustion of fossil fuels.

### 3.1.1 Willingness-to-pay unit prices

Economic value estimates for the health benefits were employed in order to make a tentative estimate of the monetised benefit from implementing the energy saving program. The unit value estimates are derived from Western studies (see US-EPA, 1995; Canadian Council of Ministers of the Environment, 1995; Krupnick *et al.*, 1996). In these studies the willingness to pay (WTP) for health risk prevention is investigated by various methods (direct or indirect), or it is estimated from the cost of illness (COI). WTP is usually higher than COI, and the WTP/COI-ratio, which is used to derive some of the unit values, has been estimated to be around 2 for many end-points (chronic diseases and mortality excluded). To estimate corresponding WTP values for Hungary we used the "relative income approach", which means using the wage ratio between the US and Hungary to adjust the WTP values. In our case the relative wage approach implies a valuation multiplier of 0.16. The unit values and estimated benefits are given in Table 2.

If we assume that WTP for health risk prevention takes an increasing share of the budget as income increases, the use of relative incomes may overstate the WTP unit prices in Hungary. On the other hand, the wage level is decisive only for a part of the COI, and it may be that other costs are relatively higher in Hungary than in the US, indicating that the relative wage income approach may understate the unit price in Hungary, if it is originally based on COI in the US. (For instance, the costs of hospital admissions and medication (embedded in the COI-part of some of the WTP-estimates) in Hungary are probably only to a limited extent a function of the wage level).

Since the end-points in our study are not exactly the same as those valued by the US-EPA (1995), we adjusted some of the estimates, and made some additional assumptions. The WTP for avoiding one case of infant death was assumed to be the same as for premature mortality in people  $\leq 65$  y. For cancer cases we used the calculation procedure proposed by the Canadian Council of Ministers of the Environment (1995), converting the estimate into US\$. The survival rate has a large impact on the estimate (a sensitivity analysis is given in Aunan, 1998). We assumed a mean 5-year survival rate of 20%; this may still be too high (see Scientific American, 1996).

To obtain unit values for impacts of respiratory symptoms, we assumed that 10% (an uncertainty interval of 5%-15% is used in low/high estimates) of the estimated acute respiratory symptom days (ARS) in Hungary are relatively severe and involve full activity restriction, i.e. a work day loss (see Aunan (1996) for a discussion of this assumption). For the end-point denoted “restricted activity days” (RAD) by US-EPA (1995) it is assumed that 20% entail full activity restriction, hence we could not use the unit price directly. Instead we estimated a modified unit value for what we called “ARS-restricted”, taking the daily wage multiplied by the WTP/COI ratio of 2 (our estimate became around 3 times higher than the RAD-value in US-EPA (1995)). In addition to this, we assumed that 0.5% (0.25%-0.75%) of our estimated ARS days involve a hospital admission (RHA), and applied the unit price proposed by EPA. For the remaining ARS days, we used the unit value given by EPA for “lower respiratory symptom days”, which are described as days where symptoms are noticeable but do not restrict normal activities.

Table 2. Unit values (willingness to pay) for health impacts (1994 US\$), and estimated annual benefit from implementation of the energy saving program in urban Hungary

End-point	Unit value western studies <sup>1</sup>			Unit value adjusted for Hungary <sup>1</sup>			Benefit Hungary mill US\$		
	Central	Low	High	Central	Low	High	Central	Low	High
Deaths>65y	3.4	1.9	6.8	0.548	0.306	1.097	261.2	103.5	663.4
Deaths≤65y	4.5	2.5	9.0	0.726	0.403	1.452	51.1	20.7	133.6
Infant deaths	4.5	2.5	9.0	0.726	0.403	1.452	24.8	2.6	182.9
Lung cancer cases	3.0	1.7	6.1	0.490	0.273	0.979	12.8	2.5	43.4
ARS-Child-mild	11 <sup>2</sup>	6 <sup>2</sup>	17 <sup>2</sup>	2 <sup>2</sup>	1 <sup>2</sup>	3 <sup>2</sup>	1.9	0.2	5.5
ARS-Child-restricted	186 <sup>2</sup>	93 <sup>2</sup>	279 <sup>2</sup>	30 <sup>2</sup>	15 <sup>2</sup>	45 <sup>2</sup>	3.6	0.2	16.2
ARS-Child-HA	0.014	0.007	0.021	0.002	0.001	0.003	13.4	0.7	61.0
Pseudo-croup-tot	574 <sup>2</sup>	473 <sup>2</sup>	675 <sup>2</sup>	93 <sup>2</sup>	76 <sup>2</sup>	109 <sup>2</sup>	0.00	0.00	0.01
ARS-Adult-mild	11 <sup>2</sup>	6 <sup>2</sup>	17 <sup>2</sup>	2 <sup>2</sup>	1 <sup>2</sup>	3 <sup>2</sup>	1.3	0.3	2.8
ARS-Adult-restricted	186 <sup>2</sup>	93 <sup>2</sup>	279 <sup>2</sup>	30 <sup>2</sup>	15 <sup>2</sup>	45 <sup>2</sup>	2.4	0.3	8.3
ARS-Adult-HA	0.014	0.007	0.021	0.002	0.001	0.003	9.1	1.0	31.1
Asthma days adults	36 <sup>2</sup>	13 <sup>2</sup>	58 <sup>2</sup>	6 <sup>2</sup>	2 <sup>2</sup>	9 <sup>2</sup>	2.6	0.01	11.3
CRS-Child	0.24	0.14	0.38	0.039	0.023	0.061	121.5	1.8	1042.4
CRS-Adult	0.24	0.14	0.38	0.039	0.023	0.061	143.0	58.2	347.1
TOTAL							648	192	2549

Notes: <sup>1</sup> Mill. US\$ unless noted; <sup>2</sup> US\$.

Concerning asthma the estimated unit price proposed by US-EPA (1995) applies to a moderate asthma day, whereas the function used to predict the response in Hungary applies to a moderate or severe asthma day. Thus, using the value directly, as we have done, is conservative. Concerning the unit value for pseudo-croup in children we assumed that one case involves an emergency room visit (unit value from Krupnick *et al.*, 1996)) and two work day losses for one parent. This COI-estimate was multiplied with a WTP/COI ratio of 2 (US-EPA, 1995). The severity of the chronic bronchitis cases in the basis study (Abbey *et al.*, 1993 and 1995) used by the EPA to estimate a unit value is probably quite similar to the chronic bronchitis cases estimated for adults in Hungary, and the regression coefficients for the function for annual TSP-level and chronic bronchitis (Abbey *et al.*, 1993) are approximately the same. Hence, we decided to use the unit value directly, additionally assuming that the value is applicable to chronic bronchitis in children as well. It is important to note that this WTP-estimate reflects the perceived welfare reduction of living with chronic bronchitis over the entire course of the illness, and is a measure of the present value of an effect which can span many years (as a minimum 3 months a year for at least two years). The adjusted unit price used to estimate the *annual* benefit of reduced chronic bronchitis rendered in Table 2 is calculated by assuming a duration of 5 years and a discount rate of 6%. A sensitivity analysis showed that the total health benefit is rather sensitive to assumptions about severity and duration of cases of chronic bronchitis (see Aunan, 1998). If we, for instance, assume 10 years instead of 5 years, the total benefit estimate falls by nearly 20%.

### 3.2 Bottom-up estimates of economic benefit of reduced health damage

The total health benefit estimated to be achieved by implementation of the energy saving program is given in Table 3. The uncertainty intervals take into account the uncertainty in the conversion of NO<sub>2</sub>-data into PM<sub>10</sub>, the 95% CI (Confidence Interval) in the regression coefficient, the uncertainty in the hypothetical baseline frequency of the effect (if this value is used), and the uncertainty in the conversion factor between various particle measure (if conversion is needed). The procedure for estimating the reduced health damage is given in Aunan *et al.* (1998). The aggregated uncertainty in the total health benefit estimate arising from uncertainties in important input parameters and variables is, however, better represented by performing Monte Carlo simulation. The overall probability distribution is shown in Table 3.

Table 3. **Estimated annual benefits and costs (mill. US\$) of implementing the energy saving program NEEIECP**

(Results from the bottom-up study.

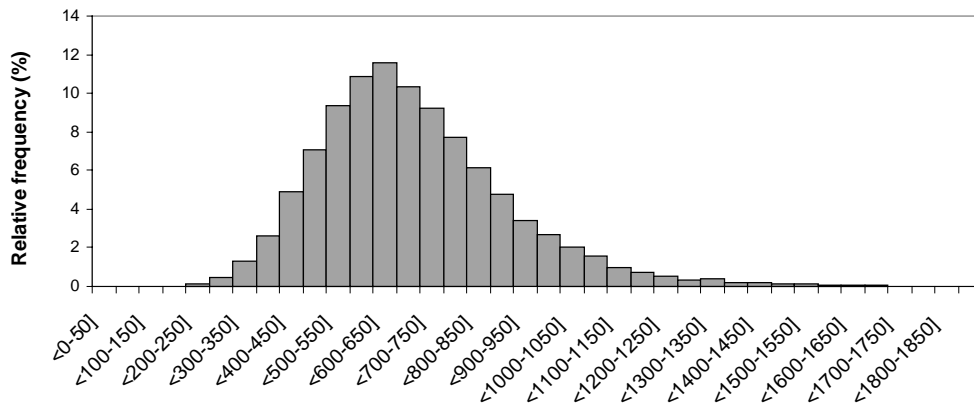
Excess mortality in people > 65 years of age is included in the health effects estimate)

	Best estimate	Range
Health effects	648	370 - 1168
Materials <sup>1</sup>	105	60 - 150
Vegetation	1.5	0.9 - 2.2
Climate <sup>2</sup>	154	86-222

1. The uncertainties are subjectively estimated to be about 40%.

2. Most estimates of the marginal CO<sub>2</sub> emission cost lie in the range 64 US\$ to 164 US\$ per tC. Our 'best estimate' is obtained using a value close to the upper limit of this interval.

Figure 1. **Probability distribution for the total health benefit of implementing the energy saving program in Hungary, as it was estimated in a bottom-up study. 10,000 trials**



The ancillary benefit per ton C, in terms of reduced health damage as it was estimated in the bottom-up study (about 482 USD/tonC), is high compared to other studies (see e.g. Krupnick and Burtraw, 1996 and Ekins, 1996). One important reason for this is that we used a separate, steeper function for mortality in people >65 years of age. This gives a considerably higher number of deaths than other studies for this group (see Table 4 in Aunan *et al.* 1997). At the same time the economic unit value for deaths (VSL) applied for this age group is quite high; 75% of the VSL for people < 65 y of age. Also, we have included a wider range of health end-points than many other studies. Moreover, the concentration level in many of the Hungarian cities is considerably higher than at least in the US and Western Europe, especially when it comes to particles. The rough approximation of assuming that a given percent overall reduction of the emissions of an air pollutant gives the same percentage reduction in the average concentration level in the cities could also have lead to an overestimation of the health benefit.

### 3.3 *Reduced material damage*

Atmospheric corrosion and deterioration of materials is a cumulative, irreversible process taking place also in the absence of pollutants. The reactivity to various air pollutants varies greatly for different materials and pollutants. Together with the level of air pollution, particularly SO<sub>2</sub> and O<sub>3</sub>, and the pH in precipitation, the deterioration processes also largely depend on meteorological conditions, especially the “time of wetness” (time fraction with relative humidity > 80% and temperature >0°C).

More knowledge about deterioration processes and better methods for assessing stock at risk have increased the possibilities for better damage assessment of building materials in recent years (see Kucera and Fitz, 1995).

We used the results from a study of the economic loss due to damage of materials in Budapest (Kruse, 1995) to assess the effect of the implementation of NEEIECP in the city. The original study used statistics of building mass and materials in the city, together with results from studies in other European cities, to assess the damage to materials. The methodology and damage functions given in Kucera and Fitz (1995) were applied, which involved estimating the reduced maintenance and replacement costs obtained from reducing the SO<sub>2</sub>-concentration level with certain steps (see details in Aunan *et al.*, 1998).

Using 1990 as the baseline year Kruse (1995) estimated the annual saving in total corrosion costs to be about US\$ 50/inhabitant, if average SO<sub>2</sub> levels were reduced to less than 20 µg/m<sup>3</sup> in all regions of Budapest. This implies a total annual saving of US\$ 100 mill. In 1990 an area representing 10-20% of the city area (in five districts in Budapest) had annual mean SO<sub>2</sub>-concentration above 20 µg/m<sup>3</sup>. We estimated that a flat 6% reduction, which is the estimated average SO<sub>2</sub>-reduction resulting from implementation of NEEIECP, should reduce the area having a SO<sub>2</sub>-level above 20 µg/m<sup>3</sup> with 20-25%. Taking into account the building density in the various districts, we arrived at a reduced annual cost in the range 30-35 mill US\$. An extrapolation of this figure to urban Hungary, using the population, gives the estimate in Table 3.

#### **4. Alternative approaches to evaluate the benefit of an improved environment**

Cost-benefit analysis of specified measures to improve the environment has traditionally been based on bottom-up assessments. As mentioned, there are alternative approaches to choose among, such as willingness to pay or cost of illness. These approaches usually turn out with widely different estimates. In this section we discuss why this is so, and suggest how both approaches may be used to do a macroeconomic analysis in order to obtain better estimates.

##### **4.1 *The reference point for alternative estimates of value of the environment***

The alternative approaches of valuation are illustrated in figure x. The thick *MC*-curve represents the marginal cost of energy savings, calculated as the cost per saved PJ of energy, and ranked according to the cost per unit of saved energy. The Energy program saves  $x(I)$  units of energy, with a marginal cost equal to  $p(I)$ . This corresponds to the unit cost of the most expensive measure, insulation and renewable energy. Without considering the environmental benefits, the usual cost benefit criterion is whether the price of energy exceeds the unit cost of the whole program, that is, whether the reduction of the electricity bill exceeds the dark shaded area. Note, however, that this is not strictly correct. If the energy price is lower than  $p(I)$ , this criteria could be satisfied, but some of the more expensive measures should not have been implemented.

One way to take environmental benefits into account is to include indirect environmental benefits from associated reduced emissions, and subtract them from the marginal costs. This leads to a negative shift in the marginal cost curve, from *MC* to the marginal social cost curve, *MSC*. Hence, the entire energy saving program might be socially beneficial, even if the alternative price of energy is lower than  $p(I)$ . In the example displayed here, inclusion of environmental benefits turns the marginal social cost of the program negative, even if the price of energy is zero. Below, we assume that the energy price is equal to zero.

An alternative approach to include environmental benefits is to consider the willingness-to-pay for improved environmental quality, for example by a questionnaire to a sample of people. An estimate of the willingness to pay ( $p(WTP)$ ) determines the point on the demand curve at which no energy saving has taken place, i.e. at  $x = 0$ . The bottom-up cost benefit criterion could be, either that  $p(WTP)$  should exceed  $p(I)$  or that the light shaded area should exceed the dark shaded area. Neither are perfect, because the willingness to pay and the marginal cost refer to different quantities of energy conservation,  $x(0)$  and  $x(I)$ , respectively. For large changes, the willingness to pay will decline as the environmental quality improves. Furthermore, the willingness to pay should be compared with  $MSC$ , to take account for reduction in damages as well.

The two approaches may yield widely different results, but none of them are correct. If energy savings actually were carried out, the willingness to pay for less pollution would decrease. Moreover, if the Energy Program is a no-regret option, it is clearly beneficial to save more energy than  $x(I)$ . Implementation of new energy saving measures would establish a new equilibrium, where the marginal cost equals the marginal willingness to pay, i.e.  $p(2)$ . This is the only point where the value of the environmental quality is defined, and the marginal social cost of the energy program is equal to the marginal willingness to pay. The amount of energy saving is then  $x(2)$ .

The critical assumptions underlying a standard bottom-up approach are, first, that average costs and benefits are interpreted as marginal costs and benefits. Second, that the use of either willingness to pay or damage cost as a proxy for the value of the environment presumes equilibrium, although the aim of the same studies frequently is to show disequilibrium. The main problem is in most cases not the use of a bottom-up approach, but rather that market effects are ignored. One solution could therefore be to carry out a partial analysis of the 'environmental market', still based on the bottom-up information, but where the marginal cost of each measure were compared with the marginal willingness to pay.

An alternative is to implement the environmental market, including the energy saving program, in a macroeconomic model. This also involves problems, because it is not possible to account for all the available information about specific measures provided by the bottom-up study. The advantage is that the relationship between the resource use of energy saving, and the economic impacts of cleaner air across the whole economy can be fully accounted for.

The macroeconomic model applied here is described in Aaheim *et al.* (2000). It was made as simple as possible in order to highlight the valuation issue. One production sector produces all commodities and services, except health services. The production sector is polluting, and delivers its output to the health sector, to households and to 'produce' abatement measures. In addition, it uses some of its output as input in own sector. Production is affected by air quality in terms of crop losses and material damage.

Households buy products from the commodity and service sector, and health services. The demand for health services was calibrated by the willingness to pay retrieved from surveys in other countries, adjusted for income level and level of air pollution. Health services are produced by means of labour and input from the commodity and service sector. The activity in the health sector is given by assumption in the case where no energy saving is carried out. To analyse the value of the Energy program, the activity in the health sector is determined by the demand for health services in households.



## 4.2 Comparing the results from the bottom-up and top-down analysis

The estimated annual benefit of the energy saving program obtained by the different approaches differ considerably, as shown in Table 4 (from Aaheim *et al.*, 2000). The bottom-up WTP estimate given in Table 4 is based on the assumption that the air pollution level in Hungary is approximately three times higher than in the US, and does not include excess mortality in people above 65 years of age (see Aaheim *et al.*, 2000 for details). The benefit estimated in the top-down analysis is, as expected, considerably lower than in the bottom-up analysis applying WTP estimates. Moreover, the estimate obtained by applying damage cost estimates typically is lower than when WTP is used, because the damage costs, such as COI, comprise market-values only. If damage cost estimates covered all damage, including the cost of the measures needed to reduce emissions, and the respondents behind a WTP estimate were 'ideal', e.g. disposing full information, one could maintain that market failure is the only reason why WTP diverged from the full marginal social cost of environmental measures. In practice, however, this is not likely to be the case. Assessment of the WTP is hampered by large difficulties (see Navrud and Pruckner, 1997). Moreover, it is difficult to make a full assessment of the damage costs, for example because necessary data are unavailable or incomplete. Symptomatically, much of the difference between the B-U estimates in the study by Aaheim *et al.* (2000) is due to the diverging estimates for chronic diseases and premature mortality, where welfare losses are likely to be more prominent than for many other end-points. Both from a practical and a theoretical point of view, it therefore seems unwarranted to apply a general WTP/COI ratio in cost-benefit analyses, as was done in the bottom-up study (Aunan *et al.*, 1998).

Table 4. **Alternative assessments of energy saving potential and environmental benefits of the Energy Saving Program**

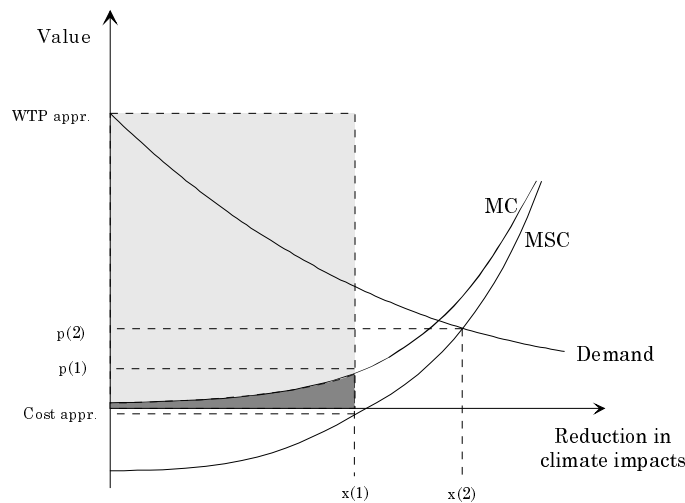
Method	Energy conservation (PJ)		Total value of ancillary benefits	Marginal value	
	Energy Program	Reduced energy use	Mill USD	Mill USD/PJ	Cents/ kWh
Bottom-up					
Damage cost	63.7	63.7	141.5	2.2	0.8
WTP	63.7	63.7	645.0	10.1	3.6
Top-down	64.5	60.0	43.5	7.3	2.6

Note:

The bottom-up damage cost estimate includes health, building materials, and agricultural crops. The bottom-up WTP estimate includes only health (see text for details). Excess mortality in people > 65 y of age is excluded in all three assessments.

The estimated market equilibrium point implies a recommendation that energy saving measures corresponding to about 65 PJ,  $x(2)$  in Figure 2, should be implemented (i.e. very close to the original program of 63.7 PJ). The defined unit value of *environmental quality*, i.e. the price of energy saving, in Figure 1, was about 7.3 mill. US\$/PJ. The corresponding total abatement cost, i.e. the integral under the cost curve in Figure 2 was 43.5 mill. US\$. These estimates are, however, very sensitive to the various assumptions made (see Aaheim *et al.*, 2000).

Figure 2. **Alternative approaches to valuation of environmental change**



## 5. Conclusions

Estimates of the value of environmental quality may depend strongly on the approach. In particular, estimates based on bottom-up studies may differ considerably according to the method. However, one should not consider alternative bottom-up approaches as ‘competing’, but rather as means to provide supplementary information for an assessment of the value of the environment. In principle, valuation ought to be assessed in a macroeconomic context, but bottom-up estimates may be appropriate when considering small changes. It may be tempting to argue that ‘small’ could be considered as changes that do not affect macroeconomic variables significantly. This may be an insufficient requirement. The results indicate that although the over-all macroeconomic effects of the Energy Program are negligible, the macroeconomic effects may be important to the values attached to the Energy Program, such as the marginal value of emission cuts and the amount of leakage due to a better environment, etc. In addition, a macro economic study provides additional information about the allocation of the environmental benefits by distinguishing between consumption and health status.

A disadvantage with a top-down approach is that a specification of measures must be expressed in general terms. For instance, the measures included in the Energy Program were expressed in terms of a cost function, which is based on a number of assumptions. The results turned out to be very sensitive to the assumptions about the shape of the cost curve for abatement measures, and to the willingness-to-pay estimate. Usually, cost curves can be established with more reliable information than was available for this study, where this proved to be difficult. Assessments of the demand for environmental qualities, such as willingness-to-pay estimates, are much more problematic. This applies in particular when based on studies in other countries, as in the present study.

The Energy Program was originally presented as a mean to reduce greenhouse gas emissions in order to make Hungary keep track with the expected commitments in an international treaty on greenhouse gas emissions. Although the value estimates of this study were based on very rudimentary information, the results indicate that the local effects on pollution probably make at least a part of the Energy Program a 'no-regret' option. This may be the case for many measures planned to reduce emissions of greenhouse gases, but local, or 'ancillary', effects of climate measures are seldom taken into account when assessing the costs of climate policy. The results here indicate that this may be a serious deficiency.

Many methodological challenges have not been discussed in this paper. One problem, related to the implementation of a cost function for energy saving measures in a macroeconomic model, is to rank the measures appropriately. In this study, the measures that constitute the Energy Program were ranked according to the unit costs per saved amount of energy. The demand for emission cuts (or energy saving) is, however, related to the demand for health services. In general, the relationship between energy saving and the effects on health varies between measures, for example because the population exposed to pollutants differs in different areas. How to do a ranking that appropriately takes these factors into account is a subject for future research.

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**ANCILLARY BENEFITS OF GHG MITIGATION IN EUROPE: SO<sub>2</sub>, NO<sub>x</sub> AND PM<sub>10</sub>  
REDUCTIONS FROM POLICIES TO MEET KYOTO TARGETS USING THE E3ME  
MODEL AND EXTERNE VALUATIONS**

by Terry BARKER, and Knut Einar ROSENDAHL

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**Abstract**

*Mitigation of greenhouse gas (GHG) emissions can lead to reductions in associated externalities, such as emissions of sulphur dioxide (SO<sub>2</sub>), fine particulate matter, and other pollutants, some of which are known to damage human health. These so-called ancillary (or secondary) benefits of mitigation policies can be contrasted with the primary benefits of such policies, namely the reduction in climate change. This paper describes how some of these benefits (associated with the reduction in 3 pollutants: SO<sub>2</sub>, oxides of nitrogen (NO<sub>x</sub>) and some fine air-borne particles (PM<sub>10</sub>)) have been assessed for 19 regions of Western Europe. The analysis uses a large-scale econometric model of Europe, E3ME, and assumes that Kyoto targets are achieved by using stylised economic instruments, with tax/permit revenues spent on reducing employer taxes.*

*The paper first explains how local and regional damage costs from emissions of NO<sub>x</sub>, SO<sub>2</sub> and PM<sub>10</sub> are included in E3ME. The damage costs are taken from the ExternE study, which is a substantial assessment of the external costs of electricity generation in Europe funded by the European Commission. Damage costs vary across pollutant and across country in the model. The projection of damage costs to 2010 shows a dramatic fall due to expectations of large target reductions in emissions of NO<sub>x</sub> and SO<sub>2</sub>. The Kyoto protocol requires that EU countries reduce GHG emissions (CO<sub>2</sub> and five other GHGs) by 8% in 2008-2012 compared to 1990 or 1995. The paper reports that since the non-CO<sub>2</sub> GHGs are projected to fall significantly over the period 1995-2010, CO<sub>2</sub> emissions have to be reduced by merely 2-3% below 1990 levels.*

*Ancillary benefits are estimated under three alternative mitigation scenarios that meet the Kyoto targets: multilateral carbon taxes, a CO<sub>2</sub> emission-permit scheme, and a combination of policies. The necessary tax rates or permit prices are 135 to 154 euros (2000) per tonne carbon. In all the scenarios, the estimated ancillary benefits by 2008-12 are about 9bn (1990) euro per year, i.e., about 138 euro (2000) per tonne reduction in carbon-equivalent (e/t) or 0.11% of total GDP. They represent, each year, a saving of around 104,000 life-years, 11,000 fewer new incidences of chronic bronchitis in adults, and 5.4 million fewer restricted activity days. These benefits constitute 15-35% of the change in GDP brought about by the mitigation policies, showing the importance of including ancillary benefits in the overall assessment of mitigation policy, even though emissions of NO<sub>x</sub> and SO<sub>2</sub> are expected to fall significantly by 2010.*

*There are three reasons why this estimate may be nearer the lower bound of the range of possible outcomes. First, if the SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> target reductions are not reached, and emission coefficients remain at 1995 levels, the ancillary benefits rise to some 300 e/t, or 0.25% of GDP. Second, if CO<sub>2</sub> emissions rise more than the modest 9% from 1990 to 2010 in the base scenario then mitigation policy will have to be stronger and the ancillary benefits will also be larger (e.g. 140 e/t or 0.34% of GDP). Third, if oil prices fall well below the assumed \$(2000) 22.5 per barrel in the baseline projection, then the tax/permit policies will again have to be stronger and the ancillary benefits could rise (to 141 e/t or 0.22% of GDP). (But if world oil prices rise to about \$(2000) 40 per barrel by 2010, then fuel use is so reduced that no additional mitigation policies are needed; and ancillary benefits are zero.) Finally, ExternE estimates are themselves conservative, covering only 3 main types of air pollutants and including only some of the damages from the pollution. Even where data exist on the pollutants, they are likely to underestimate the true size of the effect.*

## 1. Introduction

### 1.1 *Purpose of the paper*

Mitigation of greenhouse gases (GHGs), particularly CO<sub>2</sub>, can often have favourable impacts on emissions of other pollutants and on other damaging side-effects, mainly because the burning of fossil fuels is reduced. When the overall benefits and costs of climate policies are assessed, it is thus important to include such benefits e.g. those from reduced local and regional pollution. As the main benefit of climate policies is to reduce climate change, these benefits are usually referred to as ancillary benefits (or secondary benefits). Reductions in other negative externalities, especially related to road transport, are also often included in studies of ancillary benefits. Ekins (1996) gives a review of earlier European studies of ancillary benefits. Pearce (2000) reviews the current literature in the context of policy advice.

This paper develops a top-down framework for estimating the impacts on ancillary benefits of GHG mitigation policies. It is an application of a large-scale 19-region energy-environment-economy model for Europe (E3ME: see the model description in <http://www.camecon.co.uk/e3me/index.htm>). An earlier version of E3ME covering 11 regions has been applied to assess different aspects of CO<sub>2</sub> abatement, namely effects on equity (Barker and Köhler, 1998) and on competitiveness (Barker, 1998) and the advantages of EU coordination (Barker, 1999).

The valuation of emission damage is very complex, involving the connection between emissions in one set of locations at one time and pollution concentrations and exposures in other locations at later times:

- the physical impacts of pollution on human and animal health and welfare, materials, buildings and other physical capital, and vegetation; and finally
- the valuation of mortality, morbidity and other physical effects.

Much scientific effort has been devoted to the various links in this chain. Recently, several studies have been undertaken in order to estimate the costs of emissions by going through the whole chain. The most comprehensive and well-known example is the ExternE project (EC 1995), funded by the European Commission. This project was initiated to calculate external costs of electricity generation from different kind of power plants in Western Europe. The ExternE work has continued and a study of transport externalities is due to be published in 2000, but this paper relies on the results from the ExternE electricity study, rather than constructing valuations from the literature.



## 1.2 Remaining sections of the paper

In the next section a literature review is given, first on ancillary benefits and then on damage cost calculations based on the ExternE results. In section 3 the choice of methodology is discussed and the E3ME model is briefly described. Then in section 4 an assessment of current damage costs from emissions within the E3ME area is given, with a projection of damage costs to 2010 and an estimate of ancillary benefits of meeting Kyoto targets. Section 5 gives a sensitivity analysis of the results and there are some conclusions in section 6.

## 2. Literature review

Although the number of studies estimating costs of climate policies has multiplied over the last few years, particularly related to the so-called double dividend hypothesis, very few studies have emphasised or even included ancillary benefits. In Ekins' (1996) review of the literature, most reported studies are from the years 1991-93, with one exception from 1995 [e.g., Alfsen *et al.* (1992, 1995), Barker (1993) and Pearce (1992)]. Since then, the international literature contains very few such studies. One reason for this may be the difficulties in estimating such benefits, both with respect to estimating the physical effects and the corresponding economic value of specific emissions. In addition, most of these studies are site or sector specific. Thus, results cannot easily be transferred from one sector or one region to all sectors at national or international levels. The studies reported by Ekins (1996) generally use quite simplistic approaches regarding damage assessments and transferring results, and also use national estimates for the unit costs of various emissions despite the importance of location (these problems are also stressed by the authors). Moreover, in view of the epidemiological studies over the last decade, e.g. stressing the damaging effects of particulate matter, the resulting figures may be further questioned.

According to Ekins (1996) a 'consensus range' for the ancillary benefits in the studies he refers to is \$250-400 per tonne carbon reduced. All studies reviewed are from the UK or Norway. By comparing with the mitigation costs reported in the literature, he concludes that ancillary benefits alone justify large reductions in CO<sub>2</sub> emissions.

Two Norwegian studies that are not reported in Ekins (1996) are Brendemoen and Vennemo (1994) and Johnsen *et al.* (1996). However, they both use more or less the same macroeconomic model (MSG) and the same submodule for calculations of emissions, environmental damages and traffic externalities as the Norwegian studies reviewed by Ekins. Brendemoen and Vennemo find that ancillary benefits of a carbon tax make up almost all the GDP loss, and their figures indicate that these benefits amount to around \$450 per tonne carbon reduced. Johnsen *et al.* use a different baseline with much more gas power installed, and find much lower ancillary benefits, i.e., about \$70 per tonne carbon reduced, or 20% of the GDP loss. Thus, these two studies obtain results that are respectively higher and lower than the ones in Ekins (1996).

Håkonsen and Mathiesen (1997) distinguish between ancillary benefits with productive impact and benefits with direct utility impact in their model for the Norwegian economy. They find that taking ancillary benefits into account, the impact of reducing CO<sub>2</sub> emissions changes from a welfare loss of 1% to a gain of 1%.

Barker *et al.* (1993) present some ancillary benefits of a carbon/energy tax, using the macroeconometric model MDM for the UK economy. They concentrate on the traffic-related externalities, and obtain ancillary benefits around £13 per tonne carbon reduced. This is an order of magnitude below the results in Ekins (1996), due to the very small effect on petrol consumption by the tax. Moreover, benefits of reduced air pollution are not included in the study.

Bergman (1995) calculates an environmental quality adjusted national income for Sweden, using a CGE model. He takes into account ancillary benefits of reduced SO<sub>2</sub> emissions through economic welfare for households and feedback effects on production in the forest industry. With differentiated CO<sub>2</sub> taxes, the loss of gross national income is more than fully compensated by gains in environmental quality.

A newer study from Norway (Glomsrød *et al.* 1996)<sup>96</sup> employs the general method used by the ExternE project (i.e., using dose-response functions etc.) within a general equilibrium model for the Norwegian economy. Concentrations of pollutants are calculated in several towns based on emissions from various sources. Hence, the problem of site-specificity mentioned above is taken into account, as in the U.S. sector studies. Health and environmental impacts partly affect the input of the model (i.e., a simultaneous modelling of economic and environmental interactions), whereas other impacts are valued after the model is solved. Avoidable injuries associated with marginal changes in traffic are also included in the model, whereas other traffic-related effects are only assessed at the end. In this study, ancillary benefits of a gradually rising carbon tax are calculated to be 16% of the GDP loss (for the effects included in the model), half of it coming from reduction in traffic injuries. Compared to the reduction in CO<sub>2</sub> emissions, the ancillary benefits amount to about Nkr 200 per tonne CO<sub>2</sub>, or \$110 per tonne carbon. The assessment of other traffic-related benefits indicates a doubling of the ancillary benefits, i.e., still somewhat below the 'consensus range' in Ekins (1996). One reason for the small benefits of reduced air pollution is that the emissions of particulate matter in the towns, being the main contributor to health damages, are not affected very much by the carbon tax.

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The study is in Norwegian, but a brief presentation in English is given in Alfsen and Rosendahl (1996). The modelling approach is presented in detail in Rosendahl (1998).

Studies of ancillary benefits from carbon policies in the U.S. seem to find much smaller benefits than the ones reported by Ekins (1996). Burtraw and Toman (1997) review both general equilibrium studies and sector studies (i.e., electricity production) with emphasis on locational differences. However, many of the studies reviewed are difficult to interpret, as the policy tool is not targeted directly at CO<sub>2</sub> emissions only. Two general equilibrium studies with carbon taxes are referred to - Scheraga and Leary (1993) obtain ancillary benefits of \$33 per ton carbon reduction, whereas Boyd *et al.* (1995) find benefits amounting to \$40 per ton. Thus, the U.S. benefits are almost one order of magnitude below the European benefits. One explanation for the difference is the much lower population density for the US, 28 inhabitants per square kilometer, compared to 116 for the EU in 1996 (OECD, 1999).

Moreover, the electricity sector studies in the U.S. find even lower benefits. For instance, in a recent study Burtraw *et al.* (1999) calculate ancillary benefits of small, \$10 per ton, carbon taxes in the electricity sector in the U.S., taking into account the SO<sub>2</sub> cap imposed on overall electricity production. The benefits (related to NO<sub>x</sub> emissions only) amount to merely \$3 per ton carbon reduced, plus avoided SO<sub>2</sub> control costs of \$3 per ton carbon reduced. The authors argue that the sector studies include a better description of the physical effects than the general equilibrium models, as they take into account the locational characteristics. On the other hand, a general carbon tax imposed in the whole economy may cause relatively larger damage reduction per ton carbon reduced, as the emission reduction also takes place in cities.

Calculations of damage costs of air pollutants have become increasingly credible in recent years, partly due to the ExternE study among others. One important reason is that scientific knowledge of physical effects of air pollution has improved considerably. The updated knowledge of damage costs provides a good reason for new assessments of ancillary benefits of environmental policies, such as those for GHG mitigation.

Capros *et al.* (1999) use the GEM-E3 model with the ExternE valuations to measure the externalities of CO<sub>2</sub> mitigation in the EU. GEM-E3 is a general equilibrium model, treating each member state as a separate region. The ExternE valuations are implemented as fixed relationships between emissions of SO<sub>2</sub>, NO<sub>x</sub>, PM<sub>10</sub> and VOC in one country, and deposition/concentration of PM<sub>10</sub>, nitrates, sulphates, SO<sub>2</sub> and ozone in the same or another country (transport matrix). Further, a fixed damage cost is assigned to each pollutant per 1,000 persons in the affected country (based on the ExternE). Capros *et al.* (1999) do not cover the achievement of the Kyoto target, but rather a 10% reduction in CO<sub>2</sub> emissions in 2010 compared to 1990 (other GHGs are not incorporated in this version of the model). Nevertheless, it provides the most comparable set of results to those in the analysis below. They find that the damage reductions are substantial, with SO<sub>2</sub> and PM<sub>10</sub> emissions falling by around 25 and 28%, respectively, in the CO<sub>2</sub> mitigation scenarios (CO<sub>2</sub> emissions are reduced by 18% from baseline). Revenues from CO<sub>2</sub> permit auctions are recycled via reductions in labour taxes. The reduction in air pollution in the EU is valued to about 0.16% of GDP.

In our work we have relied directly on the results from case-studies under the ExternE project. Sáez and Linares (1999) present an overview of damage costs from these case-studies, which cover about 60 power plants in 14 countries, throughout Europe. The damage costs are presented for SO<sub>2</sub>, NO<sub>x</sub> and fine air-borne particulates (PM<sub>10</sub>). Costs of NO<sub>x</sub> are related to their impact via deposition of nitrates. In addition an average figure for the whole of Europe is estimated for their impact via ozone creation, which is the result of complex atmospheric chemical interactions among all the other air pollutants. The combustion plants cover all EU-countries (except Luxembourg).

A summary table is constructed in their paper for damage costs related to emissions in each EU-country, showing a wide range of values, reflecting site-specific differences. The damage costs are presented as intervals where the lower and upper limits are equal to the lowest and highest damage costs from plants in the relevant country. For half of the countries the upper limit is more than twice the lower limit for at least one of the three pollutants. This result underlines the importance of site specificity, both with regard to closeness to large cities and with regard to which way the emissions are transported (e.g. into the ocean). In France, one of the plants is located outside Paris, where the damage costs of particulate pollution from this plant are almost 10 times higher than the lower limit for France.

Despite the importance of site specificity even within a country, Sáez and Linares (1999) recommend the use of national figures in applications to other power plants whenever more advanced methods are impracticable. This is especially pertinent regarding efforts to estimate regional or global impacts tied with various mitigation policies. However, it may be questioned whether the figures may be used for other emission sources, as the effects on human exposure may depend crucially on where the emission is released. For instance, NO<sub>x</sub> emissions come chiefly from transportation, which probably leads to higher exposure than emissions from power plants. SO<sub>2</sub> and PM<sub>10</sub> emissions stem from a broader cross of sectors, including energy, industry, and transport.

The estimated damage costs include damages that occur within the whole of Europe. Hence, only a fraction of the damage costs reported occurs within the EU, and an even smaller fraction occurs in the country where the emissions are released. Still, in an EU perspective, the fraction is probably not very far from unity.

It is difficult to see what physical effects are behind the monetary damage costs in Sáez and Linares (1999). However, by using the methodology volume of ExternE (European Commission, 1999) together with two applications of the methodology (Krewitt *et al.*, 1999, and Schleisner and Nielsen, 1997), a rough estimate can be made of the physical effects.

### **3. Evaluating ancillary benefits**

#### **3.1 *The chosen method***

When choosing the appropriate methodology for assessing the costs of ancillary benefits, two main approaches are at hand. The simplest method relies on fixed damage cost coefficients on each pollutant in each region of the model, alternatively differing between emission sources within the region. This method can only be employed where coefficients are derived from other studies, e.g. results from the ExternE project (Sáez and Linares, 1999). The more sophisticated method relies on so-called impact-pathway method used by the ExternE researchers in their calculations. This includes relationships between emissions from a region (or possibly from an emission source in a region) and concentration levels in other regions, dose-response functions for health and environmental impacts, and valuation of physical effects. The latter method is more flexible and transparent and can be used to calculate damage costs brought upon individual regions. However, it requires far more information than the simple one.

Since the results from the ExternE project have the status of consensus estimates within the EU, and national figures are available in Sáez and Linares (1999), the simple method using fixed coefficients has been chosen. A possible extension in the future could however be to implement the impact-pathway described above. An intermediate position is adopted in the GEM-E3 model (Capros et al, 1999), as described in section 2 above.

In developing the broader regional estimates in this study, identical damage coefficients for each emission source have been used for each country-region in E3ME. The reason is first of all that there is no readily available information on the geographical dispersion of emissions within each region. Secondly, although the damage costs from road traffic emissions probably are higher than costs from power plant emissions (because it generally leads to higher human exposure), it is difficult to assess how much the coefficients should be increased. An ExternE study of transport externalities is expected to provide more reliable estimates in due course.

### 3.2 *The E3ME model*

A full description of the model, with extracts from the User's Manual, is available on the website <http://www.camecon.co.uk/e3me/index.htm>. E3ME is an econometric, dynamic, simulation model estimated using econometric panel-data techniques on cross-section and annual time-series data 1970-1995. It is an integrated E3 model covering 19 regions of Western Europe (the EU plus Norway and Switzerland), with an annual solution 1970-2012 allowing for lagged responses and a calibration of the solution using recent data and short-term forecasts 1996-2000. It has been designed specifically to simulate top-down E3 policies such as carbon taxes and emission permits (i.e. market-based instruments), which rely on relative price effects to influence economic activity and environmental emissions.

### 3.3 *SO<sub>2</sub>, NO<sub>x</sub>, PM<sub>10</sub> damages in E3ME*

The following general equations are included in the model:

$$D_j = d_j^{SO_2} \cdot E_j^{SO_2} + d_j^{NO_x} \cdot E_j^{NO_x} + d_j^{PM_{10}} E_j^{PM_{10}}$$

- $D_j$  denotes the total damage costs inflicted by region  $j$  on all countries in Europe (in Euro)
- $E_j^k$  denotes the total emissions of pollutant  $k$  ( $k=SO_2, NO_x, PM_{10}$ ) in region  $j$  (in tonnes)
- $d_j^k$  denotes the damage cost coefficients of pollutant  $k$  in region  $j$  (in Euro per tonne). For  $NO_x$  this coefficient include the effects through ozone, which is equal across countries

Total damage costs across the regions within the model are then:

$$D = \sum_j D_j$$

Note that these equations are used to estimate the *differences* in damage costs between various scenarios, not to estimate the total damage costs of air pollution in Europe per se. The ExternE methodology does not attempt to estimate total or average damage; only marginal damage from small changes in emission. Due to the possibilities of thresholds and to complex atmospheric interactions, the relationship between emissions and physical effects may be far from linear when emissions are reduced all the way down to zero.

Moreover, it is important to note that the estimates are of the damage costs *caused by* a specific region, not the costs *inflicted on* the region. Moreover, the damage costs include costs inflicted on areas outside the E3ME regions (i.e., other parts of Europe).

A point should be made about the selection of pollutants. Only damage costs of SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> are included in Sáez and Linares (1999), whereas the E3ME model contains several other pollutants that are relevant in the context of ancillary benefits (e.g. CO and VOC). However, the three selected are the ones for which the strongest consensus on damage valuation exists at this time.

This paper cannot fully assess the ancillary benefits of mitigation policies for two main reasons. First of all, the ExternE estimates are themselves conservative, covering only three main types of air pollutants and including only some of the damages from the pollution. Even where data exist on these three pollutants, they are likely to underestimate the true size of the effect. Thus, recent studies (Dominici, Zeger and Samet, in press) indicate that exposure misclassification in epidemiologic studies biases results toward the null hypothesis. This means that the relative risk, or size of the coefficients, obtained from air pollution epidemiology is likely to underestimate the true magnitude of the risk.

Another point should be made about reductions in other externalities. The literature review indicated that reductions in other traffic-related externalities, such as traffic congestion, noise, morbidity and mortality tied with traffic crashes, and local air pollution, may be important in estimating ancillary benefits of climate policies. Indeed a recent tri-national assessment from France, Switzerland and Austria suggests that the number of deaths tied with air pollution linked with traffic today is greater than those tied with traffic crashes alone (Sommers *et al.* 2000). In principle, the traffic-related effects of air pollution are included in our model. However, as noted in the beginning of this chapter, these effects are probably underestimated as we assign the same damage costs to traffic emissions as to power plant emissions. Moreover, as many other traffic-related externalities are not included at all in the model, the calculations underestimate the total ancillary benefits.

### 3.4 Damage cost coefficients

Table 1 shows the range of damage costs of SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> in the various countries according to the ExternE project, and is taken from Sáez and Linares (1999). In addition the effects from NO<sub>x</sub> emissions on ozone concentration are valued at 1,500 1995-Euro per tonne for each country.

Table 1. Damages of air pollutants (in euro (1995) per tonne of pollutant emitted)

Country	SO <sub>2</sub>	NO <sub>x</sub>	Particulates
Austria	9,000	16,800	16,800
Belgium	11,388-12,141	11,536-12,296	24,536-24,537
Denmark	2,990-4,216	3,280-4,728	3,390-6,666
Finland	1,027-1,486	852-1,388	1,340-2,611
France	7,500-15,300	10,800-18,000	6,100-57,000
Germany	1,800-13,688	10,945-15,100	19,500-23,415
Greece	1,978-7,832	1,240-7,798	2,014-8,278
Ireland	2,800-5,300	2,750-3,000	2,800-5,415
Italy	5,700-12,000	4,600-13,567	5,700-20,700
The Netherlands	6,205-7,581	5,480-6,085	15,006-16,830
Norway	Na	Na	Na
Portugal	4,960-5,424	5,975-6,562	5,565-6,955
Spain	4,219-9,583	4,651-12,056	4,418-20,250
Sweden	2,357-2,810	1,957-2,340	2,732-3,840
United Kingdom	6,027-10,025	5,736-9,612	8,000-22,917

Source: Sáez and Linares (1999).

Notes: (1) na: not available. (2) Although the numbers in this and later tables are shown with apparent precision, this does not indicate their accuracy. The uncertainty of the estimates and projections is discussed in the text.

As noted above, the intervals in Table 1 cover the damage costs from various power plants in the specific country. It can be seen that the differences between lower and upper limit are quite large, especially for particulates. This is because local effects are relatively more important for particulates than for the two other pollutants. As a comparison, Rosendahl (2000) finds that the local marginal costs of PM<sub>10</sub> emissions in four cities of Norway range from about 60 to 150 thousand euro (1995) per tonne (highest for Oslo). This study is based on the same methodology as ExternE, using a detailed dispersion model for each city. *These results indicate that the local damage costs of PM<sub>10</sub> emissions within cities may be much higher than the damages from power plants shown in the table above.*

There is no suitable information about how representative the plants are with respect to impact of emissions. In United Kingdom, for instance, there are case studies for three plants. The plant with the lowest damage costs of particulate emissions (i.e., 8,000 euro per tonne) is situated at the western tip of south Wales, between the sea and the mountains. The plant with the highest costs (i.e., 22,917 euro per tonne) is situated on the south coast of England, upwind of London. The third plant, with damage costs in the middle (i.e. 14,063 euro per tonne), is located in Yorkshire. It is difficult to state whether the damage costs from these three plants are representative or not for UK emissions in general. Hence, the average of the unit costs reported by the individual plants in each country has been chosen for this study. The estimated cost coefficients ( $d_j^k$ ) for damages are shown in Table 2 with the ozone-effect of  $\text{NO}_x$  emissions included. The large variance between unit costs from different plants within a country implies that the coefficients are very crude figures, and should be used with caution. The number of plant locations in each country is shown in parentheses behind the country-name, which may be an indication of how representative the coefficients are.

Table 2. **Damage cost coefficients (in 1995-Euro per tonne pollutant emitted)**

Country (no. of plants)	$\text{SO}_2$	$\text{NO}_x$	$\text{PM}_{10}$
Austria (1)	9,000	18,300	16,800
Belgium (2)	11,765	13,295	24,536
Denmark (3)	3,603	5,421	5,028
Finland (3)	1,373	2,683	1,835
France (3)	10,567	15,967	24,867
Germany (3)	12,077	14,606	21,589
Greece (4)	4,363	5,800	4,944
Ireland (2)	4,050	4,375	4,108
Italy (9)	8,688	10,007	10,400
The Netherlands (2)	6,999	7,259	16,137
Portugal (3)	5,218	7,830	6,439
Spain (13)	6,684	9,072	7,654
Sweden (2)	2,584	3,649	3,286
United Kingdom (3)	7,623	9,143	14,993

The differences in damage costs across the countries are remarkable, but reasonable. First, notice that the highest damage costs are related to emissions released in the middle of Europe, i.e., France, Belgium, Germany and Austria (see also Capros et al, 1999, ch. 8.4). These emissions will mainly be transported to densely populated areas, and consequently bring about relatively high damage to human health. Moreover, the lowest damage costs are related to emissions in the Nordic countries, Greece and Ireland, which are located in the outskirts of Europe and not upwind of other countries (such as the UK). Thus, much of these emissions will be transported to less densely populated areas and to the ocean, and therefore bring about less damage to human health. In fact, emissions of  $\text{SO}_2$ ,  $\text{NO}_x$  and  $\text{PM}_{10}$  in France are respectively 8, 12 and 14 times more costly than the corresponding emissions in Finland. This confirms the importance of site specificity. Thus, even though it has not been possible to distinguish between sites of emissions within a country, it is possible within the E3ME model to distinguish between emissions released in various broader areas of Europe.



Comparing the countries in Table 2 with the regions in the E3ME model, there are no damage cost coefficients for Norway, Switzerland and Luxembourg, and Germany and Italy have to be divided into West- and East-Germany and North- and South-Italy. At this stage there is not enough information on the location of the plants in Germany and Italy, so the same coefficients are used as for the whole country. For Italy, the damage costs are probably higher for North- than for South-Italy; for Germany the differences are probably minor. For Norway, the average of the coefficients for Denmark and Sweden is used. For Switzerland, the average of the coefficients for Austria and for one of the plants in Germany (i.e. Lauffen situated in the south) is used. For Luxembourg, the coefficients for Belgium are used. Table 3 shows the damage cost coefficients for the regions not included in Table 2.

As health effects dominate the damage cost figures, one may ask whether the figures will increase over time (in real terms) as there is generally a positive relationship between income level and valuation of specific health effects (e.g. in willingness-to-pay surveys of mortality risks). However, this is not taken into account at this stage. In a prospective study like ours, this probably leads to an underestimation of the ancillary benefits of greenhouse gas mitigation.

**Table 3. Damage cost coefficients (in euro (1995) per tonne of pollutant emitted)**

<b>E3ME Region</b>	<b>SO<sub>2</sub></b>	<b>NO<sub>x</sub></b>	<b>PM<sub>10</sub></b>
Germany (east)	12,077	14,606	21,589
Germany (west)	12,077	14,606	21,589
Italy (north)	8,688	10,007	10,400
Italy (south)	8,688	10,007	10,400
Luxembourg	11,765	13,295	24,536
Norway	3,093	4,535	4,157
Switzerland	10,850	16,537	19,326

Source: Derived from estimates in Sáez and Linares (1999).

It is important to stress the uncertainty related to the damage cost coefficients, and that several controversial assumptions are hidden in the calculations. Uncertainty relates especially to relationships between emissions and concentrations, and to physical effects of air pollution, but also to economic valuations. For instance, one has to choose how to value premature mortality due to increased pollution levels. In the ExternE calculations, the cost of premature mortality has been estimated as the value of life years lost (VLYL) rather than the value of a statistical life (VOSL). Thus, deaths of younger people are valued far more than deaths of elderly people with few years left to live. This is a very controversial issue with big implications for the results (using VOSL would probably have increased the damage costs by 50%, see AEA (1999)).

According to Sáez and Linares (1999) the damage costs are dominated by the health impacts, linked with fine-particulate-air-pollution mortality. However, they don't describe in detail how the costs are built up from mortality effects, various morbidity effects and damages to buildings, crops etc. This will of course differ between the three pollutants, but also between countries. Whereas PM<sub>10</sub> emissions only cause health damages from exposure to particulate pollution, SO<sub>2</sub> emissions bring about health damages from both SO<sub>2</sub> and sulphates (i.e., fine particulate) exposure, in addition to impacts on buildings, crops and other natural habitats. NO<sub>x</sub> emissions produce health damages from ozone and nitrates (i.e., fine particulate) exposure, but also impacts on buildings, crops etc. The physical effect is not only determined by the extra amount of emissions, but also on where the pollutants are transported, how they react in the atmosphere, and the state of the population, building stock, crops etc. in the exposed area.

Still, we are able to construct a rough estimate of some important health effects from the overall damage assessment. The reason is that the fraction of damages coming from health effects due to particulate exposure is very high for all three pollutants. For instance, Krewitt *et al.* (1999), who present damage costs from fossil electricity generation in Germany and the EU based on the ExternE methodology, find that between 96% and 101% of total damage costs are due to health effects (more than 100% means that there are positive yield effects in agriculture). For the EU as a whole the fraction is 97%. Schleisner and Nielsen (1997), who report the ExternE implementation for Denmark, find that health damages are responsible for 99% of total damages (in their case-study mainly NO<sub>x</sub> emissions are released). Moreover, in the methodology volume of ExternE (European Commission, 1999, table 8.1), the exposure response functions recommended are totally dominated by functions related to particulate exposure (i.e., PM<sub>10</sub>, PM<sub>2.5</sub>, sulphates or nitrates).<sup>97</sup> This is confirmed by the results in Schleisner and Nielsen (1997).

Even though most damage costs are coming from health effects from particulate pollution, it is not given how large share is due to mortality effects vs. morbidity effects. For instance, according to Krewitt *et al.* (1999) the fraction is 77%, whereas Schleisner and Nielsen (1997) find that the fraction is 86%. One reason may be that the mortality effects not only depend on the extra exposure, but also on the mortality rate in the population, which differ across countries. Another reason could of course be different valuation, but this is not likely given the recommendations by the ExternE.

The conclusion for this paper is that around 80% of the overall damage costs are due to mortality effects. As mentioned above, mortality is valued based on life years lost (VLYL). In the methodology volume of ExternE (European Commission, 1999), the recommended monetary value for 'chronic' mortality is 84,330 ECU (1995) (based on a 3% discount rate). Using the information in Schleisner and Nielsen (1997) about morbidity effects, the following rough estimates of physical impacts behind 1 million Euro (1995) in total damage costs can be made:

- 800,000 Euro due to mortality effects - derived from 9.5 life years lost (i.e., 84,330 Euro per life year);
- 105,000 Euro due to chronic bronchitis - derived from 1 more adult with chronic bronchitis (i.e., 105,000 Euro per adult with chronic bronchitis);

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Some notable exceptions are acute mortality effects and effects on hospital admissions from both SO<sub>2</sub> and ozone exposure.

- 37,000 Euro due to restricted activity days (RAD) - derived from 500 more RADs (i.e., 75 Euro per RAD);
- 28,000 Euro due to other morbidity effects;
- 30,000 Euro due to damages to buildings, crops and other natural habitats.

#### **4. Estimates of some ancillary benefits**

##### **4.1 *Projection of damage costs from selected emissions within the E3ME area***

A crude assessment of total damage costs from emissions of SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> within the E3ME area can now be made. As mentioned above, the damage cost coefficients in the tables above are calculated based on marginal changes in emissions, and so these coefficients cannot simply be used with the total level of emissions in the various countries. One reason for the presumable difference between the marginal and the average damage costs is the existence of thresholds, particularly with respect to health effects for some pollutants. It should be noted that the World Health Organization (1997) no longer recommends specific air quality guidelines for particulate matter as health effects have been observed at very low levels. (Note that health effects from SO<sub>2</sub> and NO<sub>x</sub> emissions are mainly due to their transmission via particulate matter.) Since the information needed to adjust the marginal damage cost coefficients is not available, the marginal coefficients are used directly in order to arrive at a very crude assessment of the total damage costs from emissions within the E3ME area. The results are not, therefore, to be treated as credible calculation of *total* damage costs in Western Europe. Nonetheless, the method provides an assessment of the *incremental* impact of mitigation policy on pollution, making the calculated values of the damage reductions more reasonable.

Table 4. Emissions (in 1,000 tonnes) and crude assessment of corresponding damage costs [billions (90) Euro] of air pollutants in the base year 1994

E3ME Region	SO <sub>2</sub>		NO <sub>x</sub>		PM <sub>10</sub>		Total damage costs
	emissions	damage costs	emissions	damage costs	emissions	damage costs	
Austria	55	0.4	183	2.9	39	0.6	<b>3.9</b>
Belgium	279	2.9	345	4.0	27	0.6	<b>7.5</b>
Denmark	157	0.5	272	1.3	14	0.1	<b>1.9</b>
Finland	111	0.1	282	0.6	72	0.1	<b>0.9</b>
France	1,013	9.5	1,831	26.0	211	4.7	<b>40.2</b>
Germany	2,998	30.3	2,042	24.9	755	13.6	<b>68.8</b>
Greece	556	1.3	358	1.1	0	-	<b>2.4</b>
Ireland	177	0.6	116	0.5	105	0.4	<b>1.5</b>
Italy	1,436	9.6	1,791	13.7	501	4.0	<b>27.3</b>
Luxembourg	13	0.1	22	0.3	0	-	<b>0.4</b>
Netherlands	146	0.9	493	3.2	38	0.5	<b>4.7</b>
Norway	34	0.1	212	0.9	24	0.1	<b>1.1</b>
Portugal	273	1.0	379	2.0	0	-	<b>3.0</b>
Spain	2,061	10.1	1,206	8.0	33	0.2	<b>18.3</b>
Sweden	74	0.2	329	1.0	48	0.1	<b>1.2</b>
Switzerland	31	0.3	140	2.0	19	0.3	<b>2.6</b>
UK	2,697	16.5	2,289	16.8	426	5.1	<b>38.5</b>
<b>Total area</b>	<b>12,111</b>	<b>84.5</b>	<b>12,290</b>	<b>109.4</b>	<b>2,312</b>	<b>30.4</b>	<b>224.3</b>

Source: E3ME project, E3ME22 C92F7BB, January 2000.

Table 4 shows the emissions of SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> in each country in the base year and the calculated damage costs. The total calculated damage costs exceed 200bn euro (1990) for the whole E3ME area. Half the costs are due to NO<sub>x</sub> emissions, whereas SO<sub>2</sub> emissions cause more than one third of the total costs. Damage costs from PM<sub>10</sub> emissions are lower. However, there are reasons to believe that these costs are underestimated, as emissions of particulate matter within the cities are more harmful than emissions from power plants (see above). Moreover, the emissions data for PM<sub>10</sub> are much more uncertain than those for NO<sub>x</sub> and SO<sub>2</sub>, and are possibly underestimated. Emissions in Germany account for more than one third of total damage costs from SO<sub>2</sub> emissions, more than one fifth of total damage costs from NO<sub>x</sub> emissions, and almost half the total damage costs from PM<sub>10</sub> emissions. This is both due to a high level of emissions and relatively high damage costs per tonne emission compared to other countries. Damage costs from emissions in France and the UK are also high; UK mainly because of high emissions level and France mainly because of high damage costs per tonne emission. Total emissions in Italy are either higher or equal to the level in France, but since damage costs per tonne emission are lower, the total damage cost for Italy is much lower than that for France. Similarly, Spain also has high emissions but low marginal damage costs.

Figure 1. **Baseline emissions 1990/95 to 2008-12: EU15 selected non-GHG pollutants**

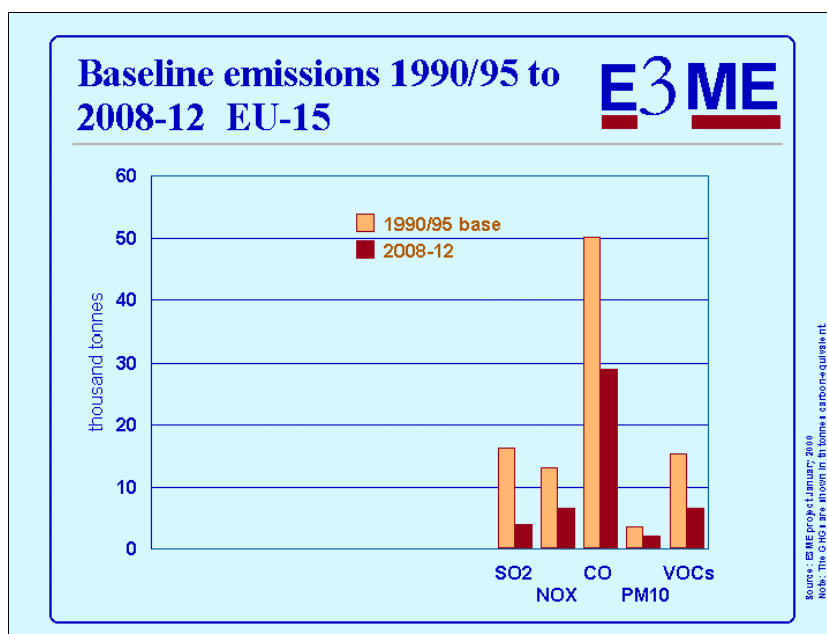


Table 5. **Crude assessment of annual damage costs (billions (90) Euro) of air pollutants in 2008-12 (baseline)**

E3ME Region	SO <sub>2</sub>	NO <sub>x</sub>	PM <sub>10</sub>	Total costs	Change from baseyear
Austria	0.3	1.6	0.6	2.5	-35%
Belgium	1.0	2.0	0.8	3.9	-48%
Denmark	0.2	0.6	0.2	1.0	-46%
Finland	0.1	0.4	0.1	0.6	-29%
France	3.9	12.3	2.7	18.8	-53%
Germany	6.1	13.1	13.2	32.5	-53%
Greece	1.3	1.1	-	2.4	-2%
Ireland	0.1	0.2	0.4	0.8	-47%
Italy	3.2	7.6	4.0	14.8	-46%
Luxembourg	0.0	0.1	-	0.2	-56%
The Netherlands	0.3	1.7	0.5	2.5	-46%
Portugal	0.6	1.4	-	2.0	-34%
Spain	2.9	5.4	0.2	8.5	-54%
Sweden	0.1	0.4	0.2	0.8	-38%
United Kingdom	4.4	9.0	3.9	17.3	-55%
<b>Total EU-15</b>	<b>24.6</b> <b>(-71%)</b>	<b>57.0</b> <b>(-46%)</b>	<b>26.8</b> <b>(-11%)</b>	<b>108.4</b>	<b>-51%</b>

Source: E3ME project, E3ME22 C92F7BB, March 2000.

Figure 1 shows the projected falls in the emission tonnages in the baseline. Note that emission coefficients of  $\text{SO}_2$  and  $\text{NO}_x$  are calibrated so that national emissions in 2010 are in accordance with a European protocol for transboundary air pollution (United Nations, 1999).<sup>98</sup>  $\text{PM}_{10}$  emission coefficients are in general supposed to follow the trend in 1990-95, as there is no protocol for this pollutant. The emissions data for  $\text{PM}_{10}$  are very uncertain, which means that this extrapolation is indeed questionable. Moreover, damage-cost coefficients are held constant.

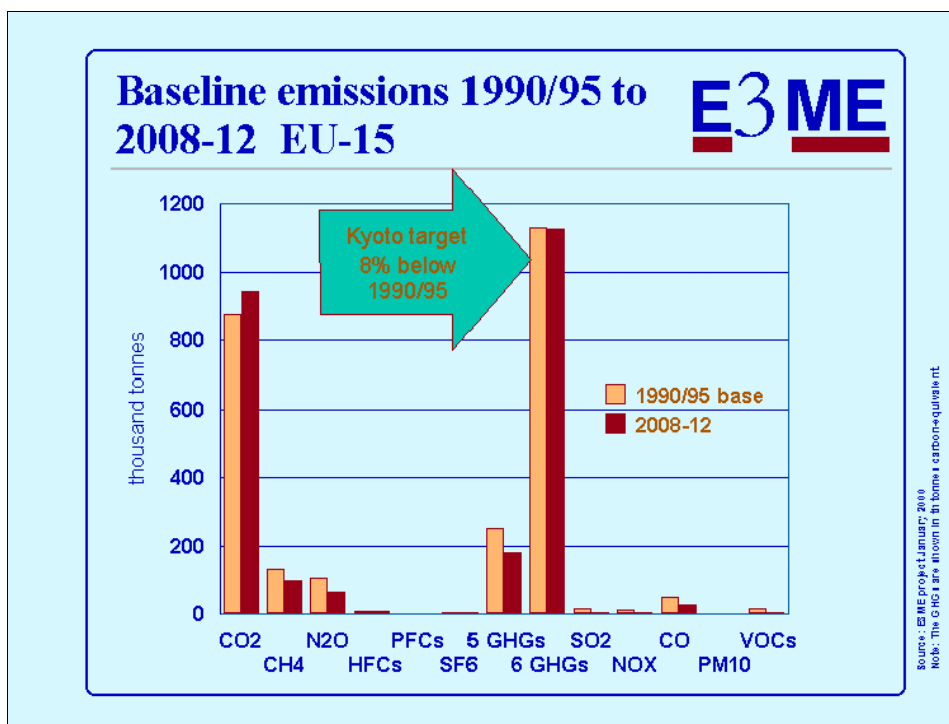
Table 5 shows the projected annual damage costs, 2008-12, for the EU-15 countries based on the E3ME baseline simulation. The total damage costs fall by 51% from 1994 to 2008-12. Because Greece is largely exempt from much of the air pollution protocol, its emissions fall the least. This overall reduction in damage costs is due to the requirement of large reductions of  $\text{SO}_2$  emissions in Europe, but also significant reductions of  $\text{NO}_x$  emissions.  $\text{PM}_{10}$  emissions fall moderately over this period, according to the model results. However, as indicated above, this last finding should be treated with great caution. The results imply that the protocol eventually brings about damage cost reductions of about 115bn euro (1990) per year compared to the base year levels. It is difficult to know how large emissions would be without the protocol: they could increase or decrease over time, due to a mix of economic growth, technological improvements and environmental controls.

$\text{NO}_x$  emissions 2008-2012 account for just above 50% of total damage costs, whereas  $\text{SO}_2$  and  $\text{PM}_{10}$  emissions account for just below 25% each. As mentioned before, total  $\text{PM}_{10}$  emissions are probably undervalued, which means that the percentage reduction in damage cost over the period will be somewhat lower. Damage costs are mostly reduced in the United Kingdom, Spain, France and Germany, where costs are more than halved. These countries have obliged to the largest reductions in  $\text{SO}_2$  and  $\text{NO}_x$  emissions compared to their emissions in 1994 (i.e., the baseyear of the model). On the other hand, costs caused by emissions in Greece are more or less unchanged.

#### 4.2 Analysis of ancillary benefits of GHG mitigation

GHG mitigation policies will lead to reductions in SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> emissions, and other air pollutants not considered in detail in this paper. Emissions of CO<sub>2</sub> and the three pollutants above are closely related to combustion of fossil fuels, so that many of the changes in energy demand and technologies resulting from GHG mitigation policies will also reduce these pollutants. In other words, actions reducing CO<sub>2</sub> emissions will indirectly reduce emissions of the other three pollutants and will reduce local and regional costs from health and environmental damages. The effects of policies to reduce other non-CO<sub>2</sub> GHG emissions are already incorporated in the baseline of the E3ME, based on projections by the IPCC (see Ellingsen *et al.*, 2000). Major reductions are projected for emissions of N<sub>2</sub>O and methane: the N<sub>2</sub>O reductions are due to strong environmental policies aimed at improving the pollution performance of combustion engines; and the methane reductions are due to no-regrets policies leading to the increased capture and use of the gas from waste disposal, coal mining and agriculture. Hence, in the baseline annual non-CO<sub>2</sub> GHG emissions are reduced by 28% in 2008-12 compared to 1990/95 as illustrated in Figure 2.

Figure 2. **Baseline emissions 1990/95 to 2008-12: GHGs and other pollutants**



In order to use the damage-cost estimates, it must be assumed that the projected levels of pollutants are not fixed by command-and-control policies (e.g. capped by the Second Sulphur Protocol). If they are capped, then the ancillary benefit from GHG mitigation will take the form of avoided costs, namely a reduction in the investment in pollution control.

Three scenarios are investigated. All the scenarios reduce the total annual GHG emissions for the EU in 2008-12 by 8% compared to the baseyear (taken as 1990 for CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O; 1995 for HFC, PFC and SF<sub>6</sub>).<sup>99</sup> No country-specific commitments apply. Permit trading is confined to the EU, Norway and Switzerland; and there is no significant contribution from joint implementation (JI) or clean development mechanism (CDM) projects in meeting EU targets. The base year value for Kyoto GHG target for the EU is 1129.9m tonne carbon-equivalent (mtC), so that an 8% reduction is 1039.5 mtC. The 1990 total for CO<sub>2</sub> is 877.0 mtC. Since non-CO<sub>2</sub> GHG emissions are reduced by 28% in the baseline, CO<sub>2</sub> emissions have to be reduced by merely 1.9% to 2.3% in the three scenarios. The base projection is denoted ‘base’ or ‘reference base’ in the tables.

There are further assumptions underlying the scenarios. Interest rates and exchange rates are held at baseline levels. The rest of the world is assumed to be unaffected by the EU achievement of Kyoto targets (i.e. the world oil price does not change from baseline levels - the sensitivity of the results to changes in oil prices is examined in the next section). Prices and wage rates are determined by estimated behavioural responses to costs and changes in market conditions. And employment can adjust freely to changes in demand. The three mitigation scenarios are as follows:

### **Mitigation Scenario 1: ‘carbon tax’**

A multilateral carbon tax

This scenario assumes that all 19 European regions and sectors (including electricity, transportation and households) are subject to the same carbon tax rate in the form of additional excise duties on energy products in proportion to their carbon content. The rate is set at 15.4 euro/toe and increased by 15.4 euro every year for the simulation period. This escalation is computed (by a trial and error procedure) to achieve a reduction in EU GHGs sufficient to meet the EU target of an 8% reduction below the 1990/1995 base (the 1995 base is chosen for the GHGs HFCs, PFCs and SF<sub>6</sub>). The electricity industry is taxed on the carbon content of its inputs, allowing for full passing on of the extra costs in the electricity prices. All revenues from such taxes are used to reduce regional employers’ contributions to social security. No permit schemes are introduced. These assumptions are chosen to approximate an ideal carbon tax in the context of an econometric simulation model, with no allowance for exemptions, WTO rules or other legal, political or social considerations.

### **Mitigation Scenario 2: ‘Permits+profits’**

In the multilateral scheme all CO<sub>2</sub> permits are grandfathered to 2000 emissions and implicit revenues are attributed to profits.

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Switzerland’s requirement is also 8%, whereas Norway’s requirement is 1% above 1990 levels. Whether or not Norway and Switzerland are included in the 8% reduction scenario has only marginal impact on the overall effects.



All regions and sectors participate in the same CO<sub>2</sub> emission permit scheme. Permit prices are endogenously determined year by year in the model by market demand and supply, and are the same across the regions. All permits are allocated on a grandfathered basis on 2000 emissions and issued to meet annual targets calculated to achieve the overall EU Kyoto target. No banking of permits is allowed and new entrants to the market have to buy permits at the market price. Target reductions for CO<sub>2</sub> permits issued to the year 2010 are calculated to be 1.9% below those of 1990 levels to achieve the 8% EU target for GHG reduction. No carbon tax schemes are introduced.

### **Mitigation Scenario 3: ‘Mixed policies’**

This is a mixed multilateral permit and tax scheme with a permit scheme for the energy sector (energy-intensive industries and electricity generation) and a carbon tax for the rest of the economy.

This scenario links energy-intensive fuel users (power generation, iron and steel, non-ferrous metals, chemicals, non-metallic mineral products and ore-extraction) in all European regions. All participate in the same CO<sub>2</sub> emission permit scheme. Permit prices are endogenously determined year by year in the model by market demand and supply, and are the same across the regions. 70% of permits are allocated on a grandfather basis on 2000 emissions in 2001, 60% in 2002, 2003 and 2004, 55% in 2005 and 50% for all later years. Reductions for CO<sub>2</sub> emissions in terms of permits issued to the year 2010 are calculated to be 25% below those of 1990 levels for the scenario to achieve the Kyoto target. All implied values of grandfathered permits are allowed to increase profits. A carbon tax at the rates in scenario 1 above is introduced for all fuel users not covered by the permit scheme, including transportation and households. All revenues from taxes and auctions are used to reduce regional employers’ contributions to social security.

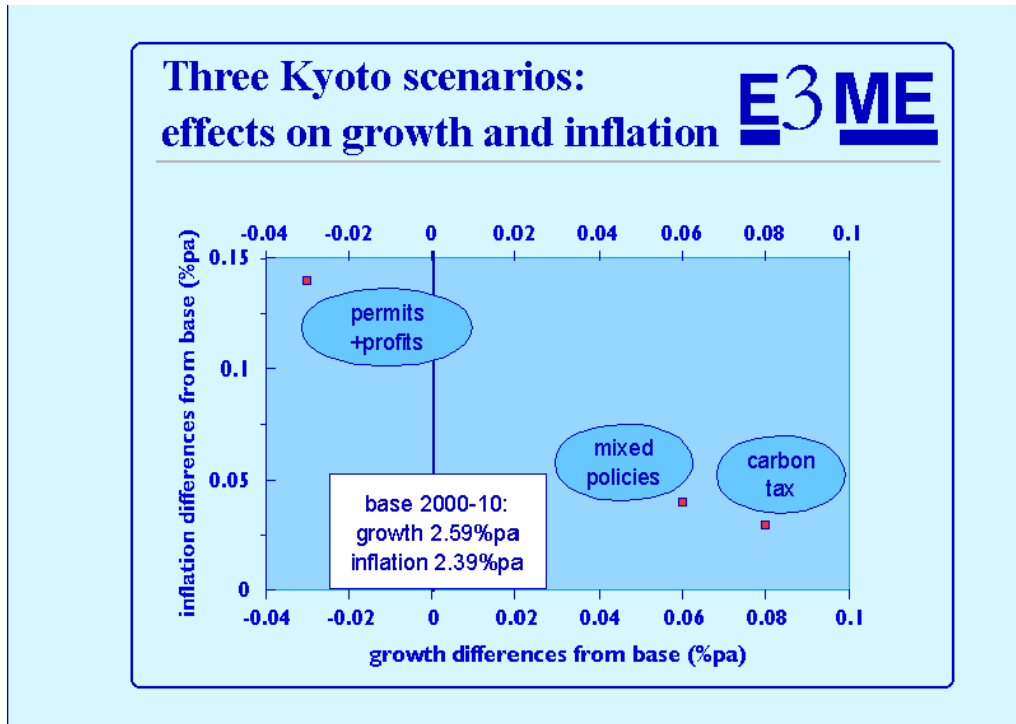
The macroeconomic results of the scenarios are discussed before those for the ancillary benefits. These are shown in Table 6 and illustrated in Figure 3.

**Table 6. Macrovariables in EURO-19 for 2010 in the three mitigation scenarios**

	<b>Base</b>	<b>Carbon Tax</b>	<b>Permits +profits</b>	<b>Mixed policies</b>
Tax rate euro(2000)/tC	0	153.1	0	153.1
Tax revenue bn euro	0	170.1	0	108.4
Permit price euro(2000)/tC	0	0	135.2	147.8
Permit revenue bn euro	0	0	0	30.7
GDP %pa 2000-10	2.6	2.7	2.6	2.6
GDP% diff from 2010 base	0	0.8	-0.3	0.5
GDP cost euro(2000)/tCe	0	-1008.5	355.7	-698.6
Anc. ben. diff. as % GDP	0	0.11	0.10	0.10
Anc. ben. euro(2000)/tCe	0	137.5	126.3	133.0
Employment 2010 m	162.2	163.9	162.1	163.5
Employ. % diff 2010 base	0	1.1	-0.1	0.8
Prices (cons.) %pa 2000-10	2.4	2.4	2.5	2.4
Prices % diff 2010 base	0	0.2	1.4	0.4
Trade bal. Pp from base	0	-0.2	0	-0.1
Gov fin bal pp from base	0	-1.2	0.2	-0.7
Energy profits bn90e dfb	0	-19.2	20.1	0.8

Source: E3ME project, E3ME22 C92F7B GHG, March 2000.

Figure 3. Kyoto scenarios: effects on growth and inflation Euro-19



According to the model, the tax rates or permit prices in 2010 lie between 135 and 154 euro (2000) per tonne carbon. Moreover, consistent with other simulations of the Energy Modelling Forum (see e.g. Weyant, 1999), the net impact on GDP is quite small, less than 1% from base in all scenarios. Indeed, in two of the three of three scenarios the GDP effect is positive. Figure 3 shows the effects on the growth rates of GPD (rather than the levels) and the rates of inflation. The effects are very small with inflation higher in all the scenarios, with the fully grandfathered permit scheme implying the highest price rises. Two scenarios increase employment by about 1% above base, whereas the second scenario, that is grandfathered permits with higher profits, shows more or less no change in employment. Introducing carbon taxes with revenue recycling seems to be the best policy choice measured in GDP and employment effects. In contrast, the permit scheme scenario with higher profits seems to be the least advantageous.

Figure 4. Effects on emissions in the mixed-policies scenario

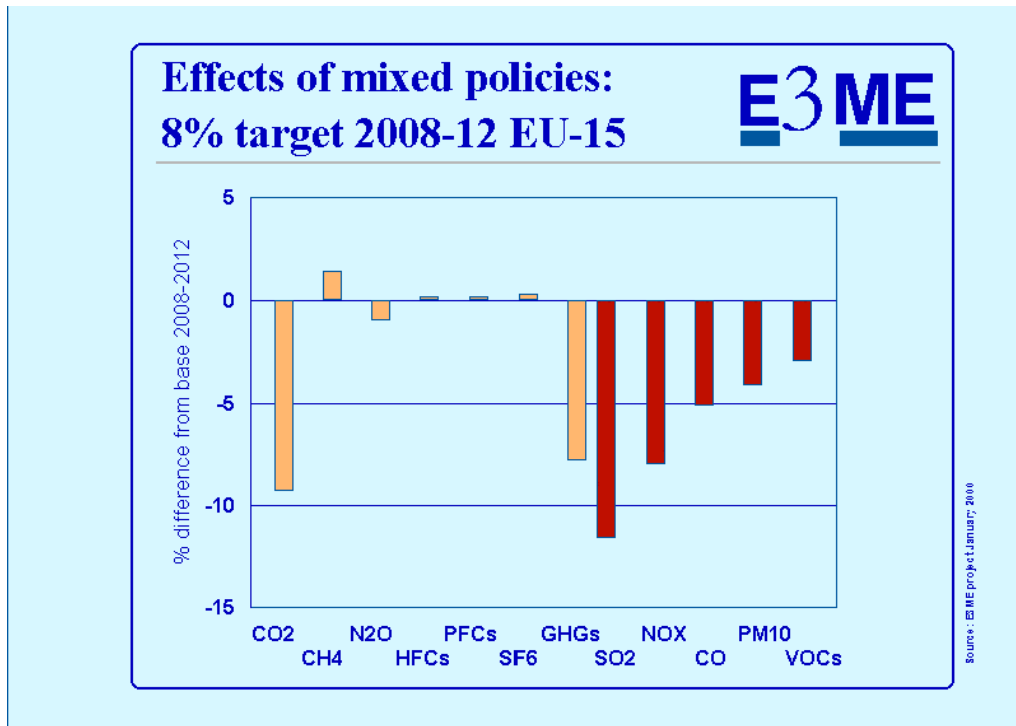


Figure 4 summarizes the overall effects on emissions in the mixed-policies scenario: here the Kyoto target is met for GHGs, with substantial further reductions below baseline in all other pollutants included in the model. Tables 7 to 9 show how much the emissions of SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> are reduced in the years 2008-12 by region in the three mitigation scenarios. The differences between the three scenarios are quite small. SO<sub>2</sub> emissions are reduced most, i.e., by around 12-13% in the EU as a whole; NO<sub>x</sub> emissions are reduced by around 8%; and PM<sub>10</sub> emissions are reduced by 4%. Moreover, the highest percentage reductions take place in Denmark (all components) and Spain (SO<sub>2</sub> and NO<sub>x</sub>). Denmark and partly Spain are also the two countries with highest percentage reduction in CO<sub>2</sub> emissions.

Table 7. Annual SO<sub>2</sub> emissions in the EU-15 over 2008-12 in the base (1,000 tonnes), and percentage change from base in the three mitigation scenarios

	Base	Carbon	Permits	Mixed
	1,000 tonne	Tax	+profits	policies
		%	%	%
Austria	39	-11.7	-10.4	-10.0
Belgium	102	-10.1	-10.0	-8.8
Denmark	58	-33.6	-29.7	-29.8
Finland	115	-10.9	-11.7	-11.5
France	410	-11.9	-11.2	-12.4
Germany	603	-11.9	-10.7	-10.7
Greece	548	-6.2	-7.0	-6.0
Ireland	41	-22.6	-19.3	-17.3
Italy	481	-20.4	-17.4	-15.8
Luxembourg	4	-8.8	-10.6	-13.3
The Netherlands	49	-0.6	-2.3	-2.5
Portugal	168	-0.3	-1.6	-0.8
Spain	592	-26.1	-23.8	-20.1
Sweden	66	-12.0	-11.5	-13.9
United Kingdom	711	-9.2	-8.2	-8.0
<b>Total EU-15</b>	<b>3,987</b>	<b>-13.5</b>	<b>-12.4</b>	<b>-11.6</b>

Source: E3ME project, E3ME22 C92F7B GHG, March 2000.

Table 8. Annual NO<sub>x</sub> emissions in the EU-15 over 2008-12 in the base (1,000 tonnes), and percentage change from base in the three mitigation scenarios

	Base	Carbon	Permits	Mixed
	1,000 tonne	Tax	+profits	policies
		%	%	%
Austria	105	-3.7	-2.9	-3.6
Belgium	175	-5.8	-5.2	-5.8
Denmark	125	-17.8	-14.7	-15.0
Finland	168	-5.4	-5.6	-6.3
France	863	-10.8	-9.3	-10.8
Germany	1,074	-7.4	-7.3	-8.0
Greece	352	-3.2	-3.7	-3.6
Ireland	64	-7.7	-6.6	-7.3
Italy	991	-7.5	-6.4	-7.2
Luxembourg	11	-4.5	-4.8	-5.8
The Netherlands	257	-5.0	-5.0	-5.6
Portugal	260	-2.6	-2.8	-3.0
Spain	809	-13.8	-12.4	-12.5
Sweden	144	-6.2	-5.5	-6.9
United Kingdom	1,220	-6.2	-5.8	-6.7
<b>Total EU-15</b>	<b>6,617</b>	<b>-7.9</b>	<b>-7.2</b>	<b>-8.0</b>

Source: E3ME project, E3ME22 C92F7B GHG, March 2000.

Table 9. Annual PM<sub>10</sub> emissions in the EU-15 over 2008-12 in the base (1,000 tonnes), and percentage change from base in the three mitigation scenarios

	Base	Carbon	Permits	Mixed
	1,000 tonne	Tax	+profits	policies
		%	%	%
Austria	39	-0.3	-0.4	-0.3
Belgium	38	-7.3	-6.6	-7.1
Denmark	49	-11.6	-9.9	-13.8
Finland	72	-4.6	-5.1	-4.8
France	120	-5.7	-4.8	-5.9
Germany	734	-7.0	-6.4	-6.8
Greece	-	-	-	-
Ireland	105	-0.6	-0.7	-0.4
Italy	498	-	-0.8	-0.1
Luxembourg	-	-	-	-
The Netherlands	37	-1.7	-1.6	-2.4
Portugal	-	-	-	-
Spain	34	-6.4	-5.3	-6.5
Sweden	80	0.1	-0.2	-
United Kingdom	325	-4.1	-3.4	-4.4
<b>Total EU-15</b>	<b>2,130</b>	<b>-4.1</b>	<b>-3.9</b>	<b>-4.1</b>

Source: E3ME project, E3ME22 C92F7B GHG, March 2000.

Table 10 shows the annual marginal change in externality damages from the three pollutants SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> in the three mitigation scenarios, assessed in terms of differences from the base level in 2008-12 (measured in billions (1990) Euro). This marginal reduction in damages constitutes the ancillary benefits from the mitigation scenarios. These benefits are of the order of 9bn euro (1990), which is a reduction in damages of slightly less than 10%. The largest benefits occur from reduced emissions in Germany, France and Spain. For Germany and France, this has to do with large initial damages in the baseline; for Spain, however, the large reduction is also related to the relatively large reductions of SO<sub>2</sub> and NO<sub>x</sub> emission relative to their baselines (see above).

Most of the benefits come from reduced NO<sub>x</sub> (50% of the benefits in the carbon tax scenario) and SO<sub>2</sub> (36%) emissions. For NO<sub>x</sub> the reason is that NO<sub>x</sub> emissions are responsible for more than half the damage costs in 2010 in the baseline, combined with a significant reduction in NO<sub>x</sub> emissions caused by the CO<sub>2</sub> tax or permit price. For SO<sub>2</sub> the reason is that SO<sub>2</sub> emissions are very responsive to a carbon tax or permit price. Whereas CO<sub>2</sub> (and NO<sub>x</sub>) emissions are reduced by just below 10%, SO<sub>2</sub> emissions are reduced by 12-13%. On the other hand, emissions of PM<sub>10</sub> are only reduced by 4%, and these damages constituted only one quarter of total damages in the base level (14% of the benefits come from reduced PM<sub>10</sub> emissions).

Table 10. **Regional externality damage (SO<sub>2</sub>+NO<sub>x</sub>+PM<sub>10</sub>). Annual average 2008-12 in billions euro (1990 prices) for base levels and differences from base**

	<b>Base</b>	<b>Carbon tax</b>	<b>Permits +profits</b>	<b>Mixed policies</b>
Germany	32.5	-2.6	-2.5	-2.6
France	18.8	-1.9	-1.7	-2.0
Spain	8.5	-1.5	-1.4	-1.3
Italy	14.8	-1.2	-1.1	-1.1
United Kingdom	17.3	-1.1	-1.0	-1.1
Rest of EU-15	16.7	-1.0	-0.9	-1.0
Eurozone EMU-11	87.0	-7.9	-7.2	-7.5
Non-EMU4	21.4	-1.5	-1.3	-1.5
<b>EU-15 (EU)</b>	<b>108.4</b>	<b>-9.4</b>	<b>-8.5</b>	<b>-9.0</b>

Source: E3ME project, E3ME22 C92F7B GHG, March 2000.

It may be interesting to compare the total ancillary benefits with the total reduction in CO<sub>2</sub> emissions due to the carbon tax or permit price. Hence, Table 11 shows the change in CO<sub>2</sub> emissions in the three mitigation scenarios. In the 'Carbon tax' scenario there is an ancillary benefit of 137.5 euro (2000) per tonne carbon reduced (see Table 6), which is slightly below the carbon tax rate. Similar results are found in the other scenarios. This benefit is somewhat below the figures from earlier European studies referred to above. One important reason for this is the projected reduction in emissions of NO<sub>x</sub> and SO<sub>2</sub> from 1994 to 2010 that will take place under planned activities that are included in the baseline. If these reductions were not to take place, the ancillary benefits would have been more than twice as high (see Section 5 below). A second reason is that the ancillary benefits include those from reduced air pollution, and not those from reduced traffic externalities, which in other studies are found to be of the same magnitude, if not larger.

Table 11. **Annual CO<sub>2</sub> emissions in the EU-15 over 2008-12 in the base (1,000 tonnes), and percentage change from base in the three mitigation scenarios**

	<b>Base</b>	<b>Carbon tax</b>	<b>Permits +profits</b>	<b>Mixed policies</b>
	1,000 tonne	%	%	%
Germany	264.0	-7.2	-7.6	-7.8
France	108.4	-11.6	-10.7	-11.8
Spain	66.6	-16.5	-14.8	-14.3
Italy	124.2	-10.7	-9.2	-9.4
United Kingdom	174.8	-7.4	-7.1	-7.6
Rest of EU-15	207.6	-9.7	-9.8	-9.8
Eurozone EMU-11	711.1	-9.9	-9.6	-9.8
Non-EMU4	234.5	-7.8	-7.5	-7.9
<b>EU-15 (EU)</b>	<b>945.6</b>	<b>-9.4</b>	<b>-9.0</b>	<b>-9.3</b>

Source: E3ME project, E3ME22 C92F7B GHG, March 2000.

The ancillary benefit per tonne reduction in carbon emissions is particularly high in countries where the population density near major energy sources is higher, such as France (196 euro), Germany (178 euro), or where the baseline emissions per capita are high, such as Spain (177 euro). In contrast, ancillary benefits are relatively low in regions where the population density is lower, such as Rest of EU-15 (65 euro), which includes the Nordic countries, Greece and Ireland. The ancillary benefits for the UK (111 euro) and Italy (117 euro) are also below the EU average, as the damage costs per emission are lower than in the more centrally-located countries, and because emissions of SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> are not reduced very much.

GDP increases in two of the three mitigation scenarios, so the environmental ancillary benefits should be added to the economic benefits of mitigation. GDP rises by at most 70bn euro (1990) and ancillary benefits add a further 14% of the change. In the case where GDP is reduced, the ancillary benefits offset about 35% of the GDP loss.

What does the size of the ancillary benefits mean for the reduction in life years lost and morbidity effects? Based on the discussion in section 3 above, the benefits by 2010 represent, each year, a saving of around 104,000 life-years, 11,000 fewer new incidences of chronic bronchitis in adults, and 5.4 million fewer restricted activity days. If traffic-related ancillary impacts were included this number would increase substantially, since the average death or injury that occurs due to traffic crashes is much younger, than that linked with air pollution. Since almost all costs are related to health damages, and almost all health damages are related to exposure to fine particles (NO<sub>x</sub> and SO<sub>2</sub> emissions can be transmitted via secondary particles), the number of lifeyears saved can approximately be distributed on the three pollutants according to their share of ancillary benefits. Consequently, about 50% of the life years saved are due to lower NO<sub>x</sub> emissions, about 36% to lower SO<sub>2</sub> emissions and about 14% to lower PM<sub>10</sub> emissions.

## **5. Sensitivity of estimated ancillary benefits to changes in assumptions**

### **5.1 *Design of the sensitivity scenarios***

The estimation of the value of the ancillary benefits of GHG mitigation is dependent on many assumptions. This section discusses how three critical assumptions in the baseline projection affect the results. The possible range of outcomes is not, however, symmetrical around the estimated value of 0.1% of GDP found for three mitigation policies analysed above. The lower bound is zero, since the benefits are attached to the policies and a range of realistic assumptions in the baseline projections can lead to the achievement of the Kyoto targets without taxes or permits. This is the case for low fuel-use scenarios, when the resulting GHG emissions fall below Kyoto targets; or when world oil prices rise so high as to achieve the same effect (calculated to be \$40.88 per barrel by 2010 in year 2000 prices). In both cases, emissions of SO<sub>2</sub> and other pollutants will fall below base, but the benefits cannot be attributed to GHG policies and therefore by definition there are no ancillary benefits.



The three sets of assumed perturbations to the baseline projection, called ‘scenarios’ below, are:

- High pollution: SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> levels are high as a results of the technical emissions coefficients being held constant at 1995 levels, ie emission intensities are at 1995 rates over the period 1995-2012, and various targets and protocols are not met. All other coefficients are left at base levels, so that the GHG emissions are the same as in the main base case (the reference base).
- High fuel use: Here fuel use is assumed to be much higher than base, such that CO<sub>2</sub> emissions 1990-2010 grow at rates similar to the IEA’s 1998 World Energy Outlook (note that the IEA assumed world oil prices at \$17(1993) per barrel 1998-2010 for their projection).
- Low oil prices: Real oil prices (Brent crude) are assumed to fall to \$15.1(2000) per barrel by 2010 instead of the \$22.5/bbl in the base.

Table 12 shows the extra damages estimated as a result of making these different assumptions. The table shows the differences of each of the perturbed bases from the reference base, discussed in section 4 above. Thus the high SO<sub>2</sub> scenario shows much higher SO<sub>2</sub> emissions as a % difference from base and higher costs of SO<sub>2</sub> damages. Table 13 shows the effects of imposing carbon taxes in the reference base and the perturbed scenarios to achieve Kyoto targets in each scenario for GHG emissions in the EU 15; the perturbed base levels are also shown to help with the interpretation of the tables. The differences between the perturbed base levels in Table 13 and the reference base levels in Table 12 are the differences shown in Table 12. Both tables show damages both in levels and differences from base. The totals for the damages (with a change in sign) correspond to the total ancillary benefits for the carbon tax scenario discussed in Section 4 above. The effects are illustrated in Figure 5 which shows the different levels of damages in the different “views of the world” and hence the different scale of the ancillary benefits when carbon taxes at different rates are introduced to meet the Kyoto target. Each pair of stacked bars represents a scenario without and with the Kyoto target, with the overall size of the bars representing the cost of pollution and the top section of the right bars representing the ancillary benefits in each scenario.

Table 12. **Sensitivity of damage estimates for the EU-15, annual average 2008-2012 (SO<sub>2</sub>, NO<sub>x</sub>, PM<sub>10</sub>)**

		Reference Base	High Pollution	High Fuel use	Low oil price
			Difference of perturbed base from reference base		
GHG	Mt carbon-equiv. And %	1126.82	0	14.5	7.08
CO <sub>2</sub>	Mt carbon and %	945.64	0	16.83	6.94
SO <sub>2</sub>	th tonnes and %	3986.92	136.22	11.32	19.16
NO <sub>x</sub>	th tonnes and %	6616.78	118.37	17.06	6.62
PM <sub>10</sub>	th tonnes and %	2129.81	26.35	7.57	2.41
SO <sub>2</sub> cost	Euro(90)bn	24.62	37.77	2.66	4.10
NO <sub>x</sub> cost	Euro(90)bn	57.01	67.08	9.85	3.41
PM <sub>10</sub> cost	Euro(90)bn	26.82	5.93	2.21	0.71
Total cost	Euro(90)bn	108.44	110.77	14.72	8.22
SO <sub>2</sub> cost	% of GDP	0.30	0.46	0.03	0.05
NO <sub>x</sub> cost	% of GDP	0.69	0.81	0.12	0.04
PM <sub>10</sub> cost	% of GDP	0.32	0.07	0.03	0.01
Total cost	% of GDP	1.31	1.34	0.17	0.10

Source: E3ME project, E3ME22 C92F7B, E, F and L GHG, March 2000.

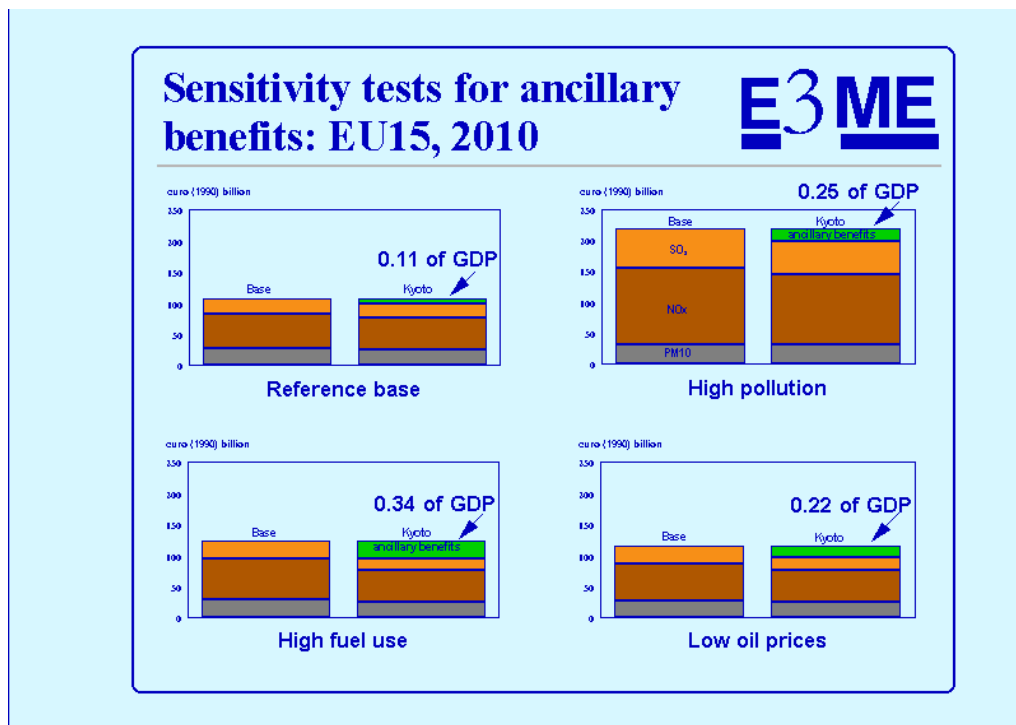
Table 13. **Sensitivity of damage estimates in achieving Kyoto targets via carbon taxes for EU-15 annual average 2008-2012 (SO<sub>2</sub>, NO<sub>x</sub>, PM<sub>10</sub>): Levels and differences from base (dfb)**

		Main case	High pollution	
		Carbon tax Dfb	Perturbed base Levels	Carbon tax dfb
GHG	Mt carbon-equiv. And %	-7.76	1126.82	-7.76
CO <sub>2</sub>	Mt carbon and %	-9.4	945.64	-9.4
SO <sub>2</sub>	th tonnes and %	-13.46	9417.88	-13.8
NO <sub>x</sub>	th tonnes and %	-7.94	14448.96	-7.9
PM <sub>10</sub>	th tonnes and %	-4.09	2691.08	-3.74
SO <sub>2</sub> cost	Euro(90)bn	-3.35	62.38	-8.51
NO <sub>x</sub> cost	Euro(90)bn	-4.68	124.09	-10.2
PM <sub>10</sub> cost	Euro(90)bn	-1.36	32.74	-1.56
Total cost	Euro(90)bn	-9.39	219.21	-20.27
SO <sub>2</sub> cost	% of GDP	-0.04	0.76	-0.1
NO <sub>x</sub> cost	% of GDP	-0.06	1.51	-0.12
PM <sub>10</sub> cost	% of GDP	-0.02	0.40	-0.02
Total cost	% of GDP	-0.11	2.67	-0.25

		High fuel use		Low oil prices	
		Perturbed base Levels	Carbon tax dfb	Perturbed base levels	Carbon tax Dfb
GHG	Mt carbon-equiv. And %	1290.24	-19.44	1206.65	-13.95
CO <sub>2</sub>	Mt carbon and %	1104.81	-23.71	1011.29	-16.94
SO <sub>2</sub>	th tonnes and %	4438.29	-25.75	4751.00	-26.19
NO <sub>x</sub>	th tonnes and %	7745.71	-22.11	7054.75	-14.61
PM <sub>10</sub>	th tonnes and %	2291.1	-11.69	2181.13	-7.09
SO <sub>2</sub> cost	Euro(90)bn	27.28	-7.42	28.72	-7.40
NO <sub>x</sub> cost	Euro(90)bn	66.86	-15.82	60.42	-9.07
PM <sub>10</sub> cost	Euro(90)bn	29.02	-4.17	27.53	-2.42
Total cost	Euro(90)bn	123.16	-27.42	116.66	-18.88
SO <sub>2</sub> cost	% of GDP	0.33	-0.09	0.34	-0.09
NO <sub>x</sub> cost	% of GDP	0.82	-0.19	0.72	-0.11
PM <sub>10</sub> cost	% of GDP	0.35	-0.05	0.33	-0.03
Total cost	% of GDP	1.51	-0.34	1.39	-0.22

Source: E3ME project, E3ME22 C92F7B, E, F and L GHG, March 2000.

Figure 5. Sensitivity tests on the ancillary benefits



## 5.2 *The high pollution scenario: higher SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> emissions*

The high pollution scenario holds the technical coefficients for SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> fixed at 1995 levels. Since the baseline allows for the effects of substantial reductions in these emissions as countries implement measures to comply with various international agreements and protocols, this means that the scenario shows much higher pollution compared with the base. However, it also shows exactly the same levels of GHG emissions, since the coefficients for these gases are left unchanged and since the emissions of SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> are assumed to have no effects on the economy or on GHG emissions. SO<sub>2</sub> and NO<sub>x</sub> emissions are expected to fall over time, even with unchanged emission coefficients, mainly as a result of changes in fuel mix away from coal and towards gas in the European energy structure. However, they are more than double the levels in the base by 2010, and the overall *extra* cost of the pollution is estimated to be of the same magnitude as the total cost in the base. The reduction in pollution, estimated to be worth an annual 1.3% of European GDP by 2010 using ExternE valuations, gives an indication of the benefits of the reduction in emission coefficients expected over the period 1995-2010, most of which can be attributed to international agreements and protocols.

The third column of Table 13 shows that if the SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> target reductions are not reached and the emission technology remains at 1995 levels, the ancillary benefits of GHG mitigation rise to 0.25% of GDP, or some 300 euros per tonne abated GHG carbon-equivalent (e/t). The first column of numbers in the table shows the corresponding effects in the main carbon tax scenario; the conclusion is that the ancillary benefits are more than doubled if pollution were to remain high. In other words, as air quality improves in Europe the value of the ancillary benefits of GHG mitigation falls substantially, although it remains significant.

## 5.3 *The high fuel use scenario*

This scenario shows the outcome if CO<sub>2</sub> emissions rise more than the modest 9% in 1990-2010 in the base scenario as a result of much higher fuel use. The fuel use in the scenario is based on that in the International Energy Agency's World Energy Outlook (1998) which is at the top of various projections of European energy use and CO<sub>2</sub> emissions (see Ybema, *et al.*, 1999, pp. 58-60 for a comparison of 6 projections to 2010). The IEA has an increase of 26% in CO<sub>2</sub>-emissions for OECD Europe 1990-2010, based on a real oil price of \$17 per barrel in 1993 prices 1998-2010. Column 4 of Table 12 shows that this scenario has 14% higher GHG emissions and 17% higher CO<sub>2</sub> emissions than the base; emissions of other gases associated with fuel use are also higher, with the overall extra cost estimated to be 0.17% of GDP.

With much higher fuel use, mitigation policy has to be much stronger (the carbon tax rate rises from 154 to 779 euro per tonne carbon), there is much more abatement, and the ancillary benefits are therefore much larger at 0.34 of GDP (see Table 13 totals). The value of the benefits is 140 e/t GHG abated. Total ancillary benefits are three times those in the reference base.

#### 5.4 The low oil price scenario

If the Kyoto Protocol comes into force, the world demand for fossil fuels will fall. Consequently, it is not unreasonable to assume that the international oil prices will fall below the assumed \$(2000) 22.5 per barrel by 2010 in the baseline projection. Then fuel use and emissions will rise and the tax/permit policies will have to be stronger to reach Kyoto targets. The scenario assumes a real oil price of \$(2000) 15.1 per barrel by 2010, about half the market price of mid-March 2000. In this scenario, GDP growth is higher and, as a consequence, electricity use is higher and coal-fired generation, which would otherwise have been closed down, is kept in operation. SO<sub>2</sub> emissions are therefore higher than in the base. Table 12 shows them to be 19% above the reference base, much more than for the other emissions. Again mitigation policy has to be much stronger than in the reference base, with the carbon tax rising to 203 euros per tonne carbon-equivalent (e/t) to achieve the Kyoto targets. Ancillary benefits rise to 0.22% of GDP or 141 e/t.

Figure 6. Sensitivity of ancillary benefits to the oil price and effects of the oil price on pollution damages

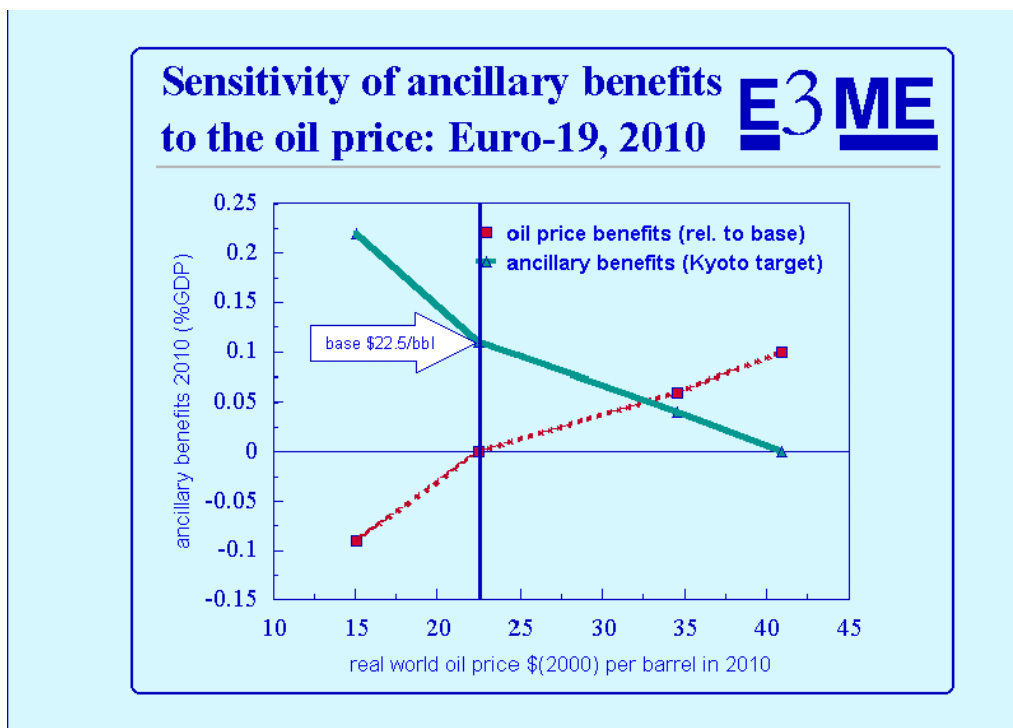


Figure 6 illustrates how the assumed level of world oil prices changes the estimated ancillary benefits. The baseline outcome is represented by the vertical line at the price of \$22.5 per barrel, year 2000 prices. At this price, ancillary benefits are 0.11% GDP, using the carbon tax to achieve the Kyoto target; by assumption the oil price effects on the pollutants are zero. When real oil prices are lower, more oil is burned and economic growth is higher; the carbon tax has to rise from 154 e/t in the base to 203 e/t for an oil price of \$15 per barrel. When oil prices are higher the carbon tax is lower and ancillary benefits diminish until at a price of \$41 per barrel (year 2000 prices) the Kyoto target is met without further policy actions and there are no ancillary benefits by definition. The dashed upward-sloping line in the chart gives the value of the changes in the SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> damages relative to the baseline which are associated with the oil price assumption. When oil prices are high, more of the reduction in damages comes from the oil price effect, and less from the carbon tax effect.

### 5.5 *Sensitivity of the ancillary benefits by region*

It is interesting to see if there are significant differences between countries in the various sensitivity tests, especially since damage costs per emission vary substantially. Table 14 shows the ancillary benefits for the achievement of the Kyoto target in the reference base and the 3 alternatives distributed on regions. Note that the benefits are attributed to the regions where the pollutants are emitted, not the regions suffering the damages. For instance, in the high pollution scenario it might be expected that unchanged emission coefficients would have increased emissions mostly in central (and northern) Europe compared to baseline, since these countries have agreed upon the highest percentage reduction in the protocol. As damage costs are highest in these regions (e.g. Germany), this would mean that damage costs should increase relatively more than emissions. However, apart from some effect for SO<sub>2</sub> costs and emissions (154% vs. 136% increase), this is not the conclusion. The reason is that much of the emission reduction is taking place through changes in fuel use and industry structure. In particular the very high sensitivity of the estimates for Spain is an effect of the use of coal in electricity generation in Spain: in the alternative scenarios there is more pollution from coal burning or much more use of coal and so more pollution.

Table 14. **Regional ancillary benefits (lower damage from SO<sub>2</sub>, NO<sub>x</sub>, PM<sub>10</sub>) – Annual average 2008-12 – Differences of carbon tax scenarios from perturbed bases as % of GDP**

	Reference Base	High Pollution	High Fuel use	Low oil price
Germany	0.14	0.31	0.43	0.25
France	0.13	0.28	0.46	0.24
Spain	0.23	0.46	0.59	0.70
Italy	0.10	0.22	0.25	0.18
United Kingdom	0.09	0.21	0.29	0.14
Rest of EU-15	0.06	0.11	0.13	0.09
Eurozone EMU-11	0.12	0.26	0.36	0.25
Non-EMU4	0.08	0.19	0.25	0.13
<b>EU-15 (EU)</b>	<b>0.11</b>	<b>0.25</b>	<b>0.34</b>	<b>0.22</b>

Source: E3ME project, E3ME22 C92F7B GHG, March 2000.

## 6. Conclusions

Most studies of GHG mitigation policies for Europe have calculated the direct costs and impacts from the use of various proposed policy instruments. To date, only one other study (Capros et al, 1999) has addressed the marginal impact of these policies for the EU on public health and the environment. The analysis above is an innovative assessment of the ancillary benefits of such policies using E3ME, a validated model of the European Community. Three different GHG mitigation scenarios are compared, each of which reaches the Kyoto target in terms of reductions in GHG emissions for the EU of 8% below 1990/1995 levels. The ancillary benefits of a carbon tax or CO<sub>2</sub> emission-permit scheme for the EU are estimated to range from zero to 0.32% of GDP depending on the assumptions chosen. Ancillary benefits here mean reductions in externality damages from SO<sub>2</sub>, NO<sub>x</sub> and PM<sub>10</sub> emissions in Europe using ExternE valuations.

As non-CO<sub>2</sub> emissions are considerably reduced in the baseline of the E3ME model, CO<sub>2</sub> emissions have to be reduced by 2-3% compared to 1990 and by 9-10% compared to the baseline projection for the years 2008-12. The three alternative mitigation scenarios use a carbon tax, a permit scheme with grandfathered permits, or a combination of these. The estimated ancillary benefits are quite similar for the three alternative scenarios; they amount to about 9 billion euro in 1990 prices, or 138 euro (2000 prices) per tonne of carbon-equivalent reduced, or 0.11% of total GDP in EU in 2010. Most of them are due to improvements in human health: these benefits by 2010 represent, each year, a saving of around 104,000 life-years, 11,000 fewer new incidences of chronic bronchitis in adults, and 5.4 million fewer restricted activity days. Moreover, these values constitute between 15 and 35% of the change in GDP (which is positive for two of the three scenarios). This means that including the ancillary benefits in the overall assessment of the policy measures is important.

The estimate of the ancillary benefits is somewhat below earlier European studies, but slightly above the results in studies from the US. One explanation for the lower benefits compared to other European studies is the much lower emissions in 2010 than in the 1990s. The importance of this was clearly confirmed by the sensitivity analysis. Another reason is that traffic-related externalities other than emissions are not included here (most American studies also include only a part of the total ancillary benefits). Even if there are uncertainties about the direct costs and benefits of GHG mitigation, the existence and scale of the ancillary benefits imply that higher direct costs of mitigation can be justified.

Around 50% of the ancillary benefits are due to reduced  $\text{NO}_x$  emissions, about 35% to reduced  $\text{SO}_2$  emissions, and about 15% to reduced  $\text{PM}_{10}$  emissions. This holds true even though the baseline projection includes large reductions in emissions of  $\text{NO}_x$  and  $\text{SO}_2$  after the signing of a European protocol on transboundary pollution. If this agreement is not effective, and emissions are not reduced as much as expected, the ancillary benefits of a carbon tax will be higher. On the other hand, measures to mitigate  $\text{CO}_2$  emissions can be a deliberate policy to reduce emissions of the other pollutants. If so, the ancillary benefits will not be related to lower damage costs, but to lower control costs. These will probably be at least as high as the damage costs, as the marginal costs at high levels of abatement (which will then be avoided) are expected to be much higher than the average costs.



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# ECONOMIC EVALUATION OF HEALTH IMPACTS DUE TO ROAD TRAFFIC-RELATED AIR POLLUTION

## An impact assessment project of Austria, France and Switzerland

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### Summary

*In preparation for the Transport, Environment and Health Session of the WHO Ministerial Conference on Environment and Health in London (June 1999) a tri-lateral project was carried out by Austria, France and Switzerland.*

*The project assessed the health costs of road-traffic related air pollution in the three countries using a common methodological framework.*

*Based on the average yearly population exposure to particulate matter with an aerodynamic diameter of less than 10  $\mu\text{m}$  (PM10) and the exposure-response function for a number of different health outcomes, the number of cases attributable to (road traffic-related) air pollution was estimated.*

*Using the willingness-to-pay as a common methodological framework for the monetary valuation, material costs such as medical costs and loss of production or consumption as well as the intangible costs of pain, suffering, grief and loss in life quality were considered. The monetary valuation provided the following results (see Summary Table).*

*All three countries together bear some 49'700 million EUR<sup>100</sup> of air pollution related health costs, of which some 26'700 million EUR are road-traffic related. In each country, the mortality costs are predominant, amounting to more than 70 %.*

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<sup>100</sup>

1 EUR  $\approx$  0.94 US \$, April 2000

The annual national per capita costs of total air pollution related health effects result in a similar range of values for all three countries. Considering the per capita health costs due to road traffic-related air pollution, the differences between the countries are even lower with a range from 180-540 EUR for Austria (central value 360 EUR), 190-560 EUR for France (central value 370 EUR) and 160-470 EUR for Switzerland (central value 304 EUR).

Summary Table. **Health costs due to road traffic-related air pollution in Austria, France and Switzerland based on the willingness-to-pay approach (1996)**

	Austria		France		Switzerland	
	Total costs with road traffic share	Costs attributable to road	Total costs with road traffic share	Costs attributable to road	Total costs with road traffic share	Costs attributable to road
Costs of mortality (million EUR)	5'000 3'000 - 7'000	2'200 1'300 - 3'000	28'500 17'300 - 39'900	15'900 9'600 - 22'200	3'000 1'800 - 4'200	1'600 1'000 - 2'200
Costs of morbidity (million EUR)	1'700 400 - 3'000	700 200 - 1'300	10'300 2'800 - 18'500	5'700 1'500 - 10'300	1'200 300 - 2'100	600 200 - 1'100
Total costs (million EUR)	6'700 3'400 - 10'000	2'900 1'500 - 4'300	38'800 20'100 - 58'400	21'600 11'100 - 32'500	4'200 2'100 - 6'300	2'200 1'200 - 3'300

	all three countries	
	Total costs with road traffic share	Costs attributable to road
Costs of mortality (million EUR)	36'500 22'100 - 51'100	19'600 11'900 - 27'500
Costs of morbidity (million EUR)	13'200 3'500 - 23'700	7'100 1'900 - 12'800
Total costs (million EUR)	49'700 25'600 - 74'900	26'700 13'700 - 40'200

## 1. Introduction

The objective of this tri-lateral research project was to quantify the health costs due to road traffic-related air pollution. The project was carried out by Austria, France and Switzerland. The results of this co-operation provided an input for the WHO Ministerial Conference in June 1999.<sup>101</sup>

The monetary evaluation of the health costs is based on an interdisciplinary co-operation in the fields of air pollution, epidemiology and economy. Figure 1 presents an overview of the different tasks of the three domains.

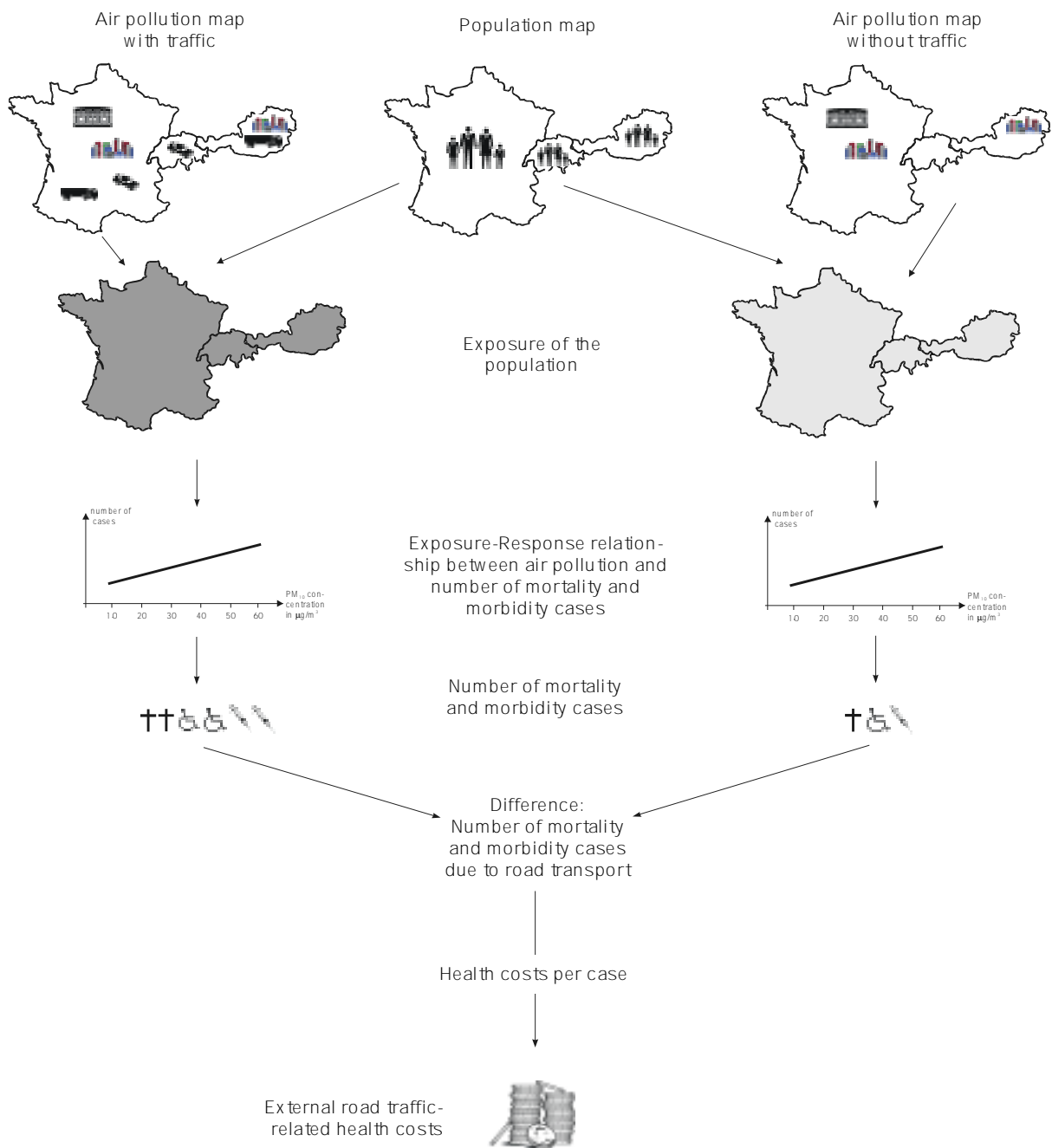
- **Air pollution:** Evaluation of the (traffic related) exposure to particulate matter: The starting point of the study is the determination of the pollution level in 1996 to which the population was exposed. The entire population of Austria, France and Switzerland is subdivided into categories of exposure to different classes of pollution levels from a superposition of the mapping of ambient concentration of particulate matter (average annual  $PM_{10}$ ) with the population distribution map. In addition, a scenario without road traffic-related emissions is calculated and the exposure under these theoretic conditions is estimated.
- **Epidemiology:** Evaluation of the exposure-response function between air pollution and health impacts: The relationship between air pollution and health has to be assessed. Thereby it has to be shown, to which extent different levels of air pollution affect a population's morbidity and mortality. This evaluation is based on the latest scientific state of the art presented in the epidemiologic literature and comprehends the results of extensive cohort studies as well as time series studies.
- **Economics:** Evaluation of the traffic-related health impacts and their monetarisation: Using epidemiological data regarding the relation between air pollution and morbidity and premature mortality, the number of cases of morbidity and/or premature mortality attributed to air pollution is determined for each of the health outcomes separately, using specific exposure-response functions. The same operations are carried out for the theoretical situation in which there is no road traffic-related air pollution. The difference between the results of these two calculations corresponds to the cases of morbidity and premature mortality due to road traffic-related air pollution. The morbidity and mortality costs arising from road traffic-related air pollution are then evaluated for each health outcome separately by multiplication of the number of cases with the respective cost estimates (willingness-to-pay factors for the reduction of the different health risks).

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Third WHO Ministerial Conference on Environment and Health, London, 16-18 June 1999.

**Figure 1. Methodological approach for the evaluation of mortality and morbidity due to road traffic-related air pollution**



Throughout the entire project many assumptions and methodological decisions had to be made along the various calculation steps in the domains of air pollution, epidemiology and economics. On each level, the method of dealing with uncertainty had to be defined. The research group decided that the main calculation ought to apply an **“at least” approach**, thus consistently selecting methodological assumptions in a way to get a result which may be expected to be “at least” attributable to air pollution. Accordingly, the overall impact of air pollution is expected to be greater than the final estimates. To unambiguously communicate the uncertainty in the common methodological framework, the final results will be reported as a range of impacts rather than as an exact point estimate.

## 2. Epidemiology - the air pollution attributable health effects

In the last 10-20 years epidemiology has dealt extensively with the effect of outdoor air pollution on human health. A considerable number of case studies in different countries and under different exposure situations have confirmed that air pollution is one of various risk-factors for morbidity and mortality.

In general, air pollution is a mixture of many substances (particulates, nitrogen oxides, sulfur dioxides). Knowing that several indicators of exposure (eg. NO<sub>2</sub>, CO, PM<sub>10</sub>, TSP etc.) are often highly correlated, it is not accurate to establish the health impact by a pollutant-by-pollutant assessment, because this would lead to a grossly overestimation of the health impact. The objective is therefore to cover as best as possible the complex mixture of air pollution with one key indicator. Based on various epidemiological studies, in the present study PM<sub>10</sub> (particulate matter with an aerodynamic diameter of less than 10 µm) is considered to be a useful indicator for measuring the impact of several sources of outdoor air pollution on human health. The derivation of air pollution attributable cases has been described in a separate publication.<sup>102</sup> Thus, the key features of the epidemiology based assessment are only summarized.

For the assessment of the health costs it was not possible to consider all health outcomes found to be associated with air pollution. Only those meeting the following three criteria were considered:

- there is epidemiological evidence that the selected health outcomes are linked to air pollution;
- the selected health outcomes are sufficiently different from each other so as to avoid double counting of the resulting health costs (separate ICD<sup>103</sup> codes);
- the selected health outcomes can be expressed in financial terms.

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<sup>102</sup> Künzli N. et al (2000), Public Health Impact of Outdoor and Traffic-related Air Pollution: A Tri-national European Assessment, in press.

<sup>103</sup> ICD: International Classification of Diseases.



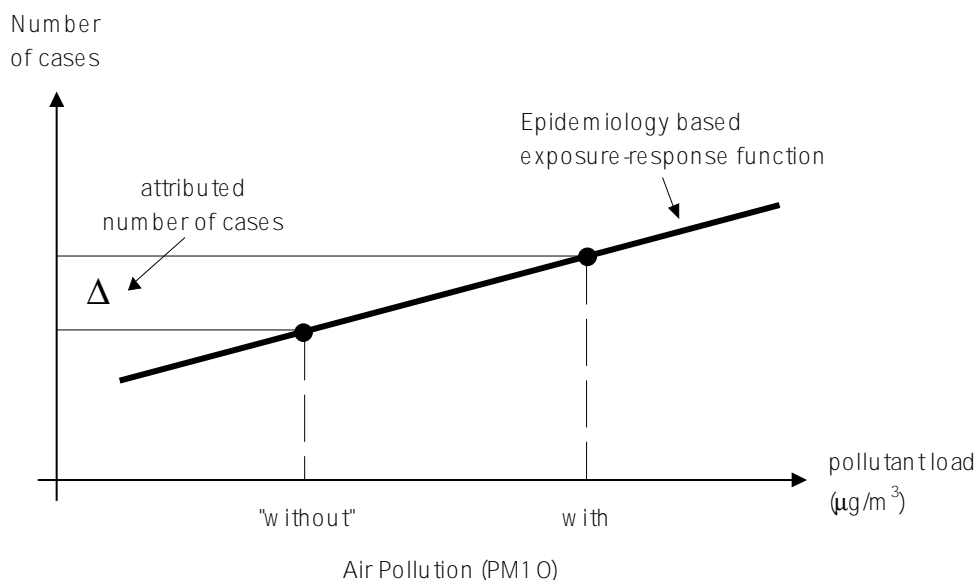
According to these selection criteria seven health outcomes were considered in this study (see Table 2).

**Table 2. Air pollution related health outcomes considered**

Health outcome	Age
Total mortality	Adults, $\geq 30$ years of age
Respiratory hospital admissions	All ages
Cardiovascular hospital admissions	All ages
Acute bronchitis	Children, $< 15$ years of age
Restricted activity days	Adults, $\geq 30$ years of age
Asthmatics: asthma attacks	Children, $< 15$ years of age; Adults, $\geq 15$ years of age

The relation between exposure to air pollution and the frequency of health outcome is presented in Figure 3 by graphical means. The number of mortality and morbidity cases due to air pollution can be determined if the profile of the curve (**exposure-response function**) and its position (**health outcome frequency**) are known. These two parameters were determined for each health outcome, separately.

**Figure 3. Relation between air pollution exposure and cases of disease**



The **exposure-response function** (quantitative variation of a health outcome per unit of pollutant load) was derived by a meta-analytical assessment of various (international) studies selected from the peer-reviewed epidemiological literature. The effect estimate (gradient) was calculated as the variance weighted average across the results of all studies included in the meta-analysis.

In this project, the impact of air pollution on mortality is based on the long-term effect. This approach is chosen because the impact of air pollution is a combination of acute short-term as well as cumulative long-term effects. For example, lifetime air pollution exposure may lead to recurrent injury and, in the long term, cause chronic morbidity and, as a consequence, reduce life expectancy. In these cases, the occurrence of death may not be associated with the air pollution exposure on a particular day (short-term effect) but rather with the course of the chronic morbidity, leading to shortening in life.

Accordingly, for the purpose of impact assessment, it was decided not to use response functions from daily mortality time-series studies to estimate the excess annual mortality but the **change in the long-term mortality rates** associated with ambient air pollution.<sup>104</sup>

Contrary to the exposure function which is assumed to be the same for all countries, the **health outcome frequency** (frequency with which a health outcome appears in the population for a defined time span) may differ across countries. These differences may result from a different age structure or from other factors (i.e. drinking and eating habits, different health care systems in the three countries, etc.). Therefore national or European data were used whenever possible to establish the countries' specific health outcome frequency.

For each health outcome included in the trinational study, Table 4 presents the effect estimates in terms of relative risks (column 2) and separately for each country the health outcome frequency (column 3-5), and the attributable number of cases for 10  $\mu\text{g}/\text{m}^3$   $\text{PM}_{10}$  increment.

*Reading example:*

The relative risk of long-term mortality for a 10  $\mu\text{g}/\text{m}^3$   $\text{PM}_{10}$  increment is 1.043 (column 2), therefore the number of premature fatalities increases by 4.3% for every 10  $\mu\text{g}/\text{m}^3$   $\text{PM}_{10}$  increment. Column 5 shows the number of deaths (adults  $\geq 30$  years) per 1 million inhabitants in Switzerland (8'260). With an average  $\text{PM}_{10}$  concentration of 7.5  $\mu\text{g}/\text{m}^3$  a baseline frequency of 7'794 deaths would be expected. This proportion depends on the age structure of the population  $\geq 30$  years and therefore is different for each country.

The absolute number of fatalities (340 cases for Switzerland, column 8) per 10  $\mu\text{g}/\text{m}^3$   $\text{PM}_{10}$  increment and per 1 million inhabitants corresponds to the 4.3% increase in mortality (column 2) applied to the baseline frequency of 7'794 deaths.

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<sup>104</sup> Künzli N. et al (2000), Public Health Impact of Outdoor and Traffic-related Air Pollution: A Tri-national European Assessment, in press.

Table 4. Additional cases per 1 million inhabitants and 10 µg/m<sup>3</sup> PM<sub>10</sub> increment<sup>105</sup>

	Effect estimate relative risk (± 95% confidence interval)	Observed population frequency, P <sub>o</sub> Per 1 million inhabitants and per annum			Fixed baseline increment per 10 µg/m <sup>3</sup> PM <sub>10</sub> and 1 million inhabitants (± 95% confidence interval)		
		Austria	France	Switzerland	Austria	France	Switzerland
Long-term mortality (adults ≥ 30 years: excluding violent death)	1.043 (1.026-1.061)	9'330	8'390	8'260	370 (230-520)	340 (210-480)	340 (200-470)
Respiratory hospital admissions (all ages)	1.0131 (1.001-1.025)	17'830	11'550	10'300	230 (20-430)	150 (20-280)	130 (10-250)
Cardiovascular hospital admissions (all ages)	1.0125 (1.007-1.019)	36'790	17'270	24'640	450 (230-670)	210 (110-320)	300 (160-450)
Chronic bronchitis incidence (adults ≥ 25 years)	1.098 (1.009-1.194)	4'990	4'660	5'010	410 (40-820)	390 (40-780)	430 (40-860)
Bronchitis (children < 15 years)	1.306 (1.135-1.502)	16'370	23'530	21'550	3'200 (1'410-5'770)	4'830 (2'130-8'730)	4'620 (2'040-8'350)
Restricted activity days (adults ≥ 20 years) <sup>a</sup>	1.094 (1.079-1.109)	2'597'300	3'221'200	3'373'000	208'400 (175'400-241'800)	263'700 (222'000-306'000)	281'000 (236'500-326'000)
Asthmatics: asthma attacks (children < 15 years) <sup>b</sup>	1.044 (1.027-1.062)	56'700	62'800	57'500	2'330 (1'430-3'230)	2'600 (1'600-3'620)	2'400 (1'480-3'340)
Asthmatics: asthma attacks (adults ≥ 15 years) <sup>b</sup>	1.039 (1.019-1.059)	173'400	169'500	172'900	6'280 (3'060-9'560)	6'190 (3'020-9'430)	6'370 (3'100-9'700)

a: Restricted activity days: total person-days per year  
b: Asthma attacks: total person-days per year with asthma attacks  
P : Frequency as observed at the current level of air pollution

### 3. Air Pollution - the PM<sub>10</sub> population exposure

In addition to the epidemiological data need, information on the population's exposure to PM<sub>10</sub> is a further key element for the assessment of air pollution-related health effects. Information about the sources and the spatial distribution of PM<sub>10</sub> is still sparse in Austria, France and Switzerland as it is in many other European countries. Therefore it was necessary to calculate the spatial distribution of PM<sub>10</sub> by using empirical dispersion models or statistical methods. The general methodological framework for the air pollution assessment consisted of four main steps:

- acquisition and analysis of the available data on ambient concentration of particulate matter (Black Smoke BS, Total Suspended Particulate TSP and PM<sub>10</sub>) for model comparison or correlation analysis between different particle measurement methods
  - PM<sub>10</sub> mapping by spatial interpolation with statistical methods or empirical dispersion modelling;
  - estimation of the road traffic-related part of PM<sub>10</sub> (based on emission inventories for primary particles and for the precursors of secondary particles);
  - estimation of the population exposure from a superposition of the PM<sub>10</sub> map on the population distribution map.

The differences between the countries concerning the procedures for measuring ambient particulate matter and the availability of emission data led to an adaptation of the general framework to the individual country specific case.

In **Austria**, particulate matter is measured in agreement with national legislation as Total Suspended Particulate (TSP) at more than 110 sites, whereas PM<sub>10</sub> measurements are not yet available. It was assumed that ambient air TSP levels can be attributed to the contribution of local sources and regional background concentrations. Both of them were modelled separately. The starting point for the modelling of local contributions was the availability of a spatially disaggregated emission inventory for nitrogen oxides (NO<sub>x</sub>). An empirical dispersion model was established for NO<sub>x</sub> whose results could be compared with an extended network of NO<sub>x</sub> monitors. The spatial distribution of NO<sub>x</sub> was converted into TSP concentrations, using source specific TSP/NO<sub>x</sub> conversion factors. The regional background TSP levels were estimated from measurements and superimposed on the contributions from local sources. These results were compared to measured TSP data. Finally, PM<sub>10</sub> concentrations were derived from TSP values by applying source specific TSP/PM<sub>10</sub> conversion factors. The model is able to provide an estimate of the traffic-related part of PM<sub>10</sub> concentration.

The **French work** was based on the available Black Smoke (BS) data. A correlation analysis between BS and PM<sub>10</sub> (TEOM method<sup>106</sup>) was first carried out. It was found that at urban background sites, BS and PM<sub>10</sub> (TEOM) are about equal. Following this, linear relationships were sought between the BS data and land use categories in the areas surrounding the measurement sites. Multiple regression analysis was performed for three categories of sites: urban, suburban and rural. Based on these regressions and using the land use data set, a PM<sub>10</sub> map was established. A correction factor for secondary particles was defined using the European scale EMEP<sup>107</sup> model. This was necessary because BS and TEOM considerably underestimate the amount of secondary particles in PM<sub>10</sub>. The percentage of PM<sub>10</sub> caused by road traffic was determined in each grid cell using results from the Swiss PM<sub>10</sub> model.

The **Swiss work** was based on a provisional national PM<sub>10</sub> emission inventory. It was first disaggregated to a km<sup>2</sup> grid. Dispersion functions for primary PM<sub>10</sub> emission were defined in an empirical dispersion model which was used to calculate the concentration of primary PM<sub>10</sub>. The contribution of secondary particles was modelled by using simple relationships between precursor and particle concentration. The long-range transported fraction was taken from European scale models. The PM<sub>10</sub> fractions were then summed to create the PM<sub>10</sub> map. The traffic related part was modelled separately, using both the road-traffic related portion of PM<sub>10</sub> emission and the respective portion of the precursor emission for secondary particles.

The determination of the **regional PM<sub>10</sub> background** was critical to the PM<sub>10</sub> mapping procedures. The estimates of all three countries are in line with measured and modelled data from EMEP. The large-scale transported fraction of PM<sub>10</sub> is considerable. At rural sites, over 50 % of PM<sub>10</sub> may originate from large-scale transport. Furthermore, the contribution of traffic to PM<sub>10</sub> background concentration is substantial and it may vary in space.

The **population exposure to total PM<sub>10</sub>** is presented in Figure 5. Around 50% of the population live in areas with PM<sub>10</sub> values between 20 and 30 µg/m<sup>3</sup> (annual mean). About one third is living in areas with values below 20 µg/m<sup>3</sup>. The rest is exposed to PM<sub>10</sub> concentrations above 30 µg/m<sup>3</sup>. The high concentrations are found exclusively in large agglomerations.

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<sup>106</sup> TEOM: Tapered element oscillating microbalance. Method for measuring continuously particle concentration.

<sup>107</sup> EMEP: Co-operative Programme for the Monitoring and Evaluation of Long-Range Air Pollutants in Europe.

Figure 5. Frequency distribution of total PM<sub>10</sub> population exposure (with share attributable to road traffic)<sup>108</sup>

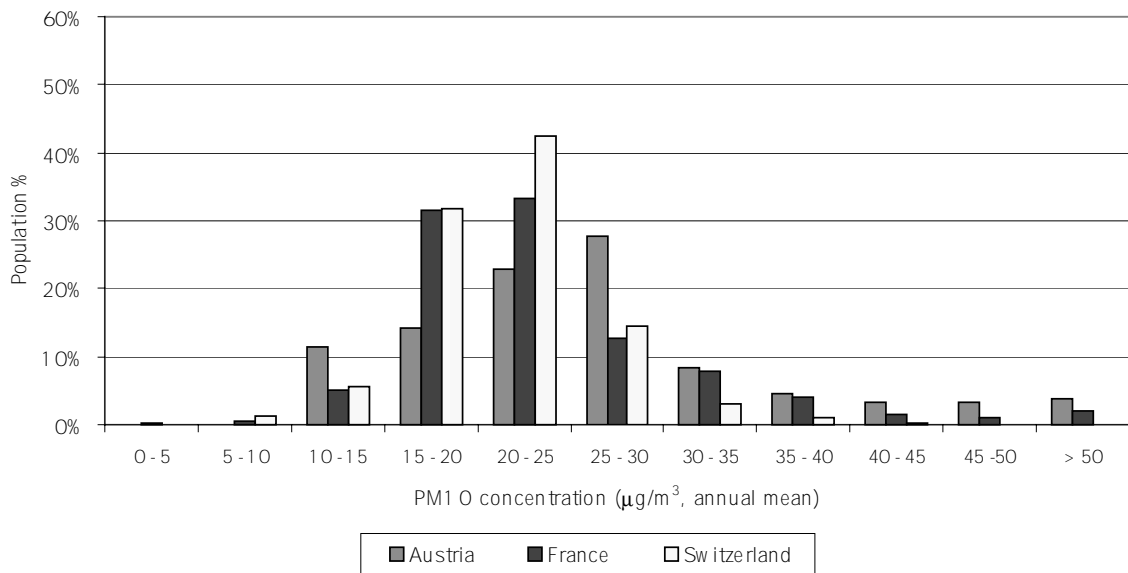
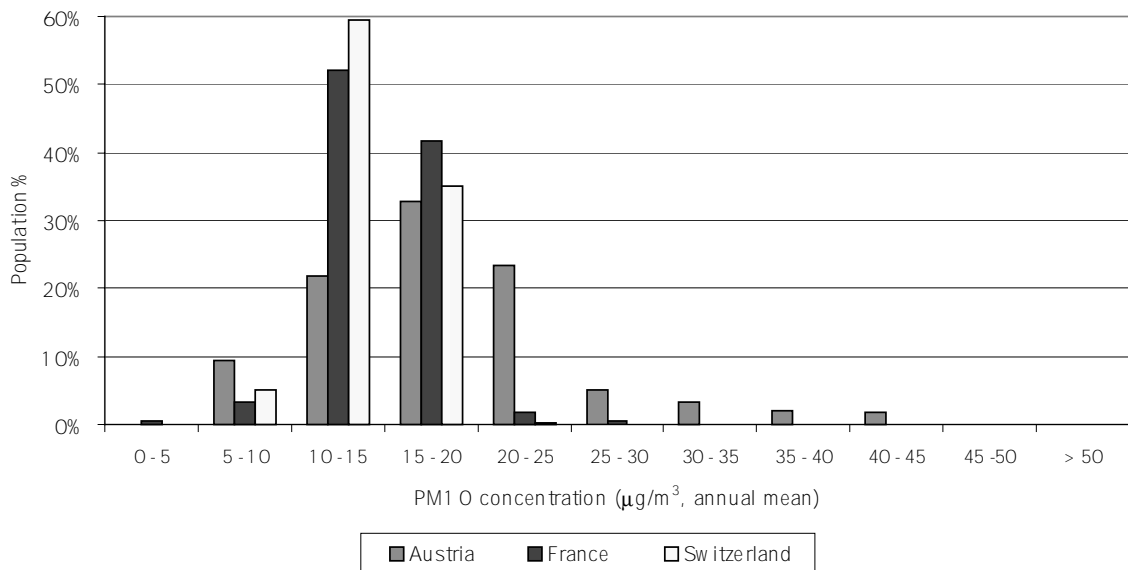


Figure 6. Frequency distribution of PM<sub>10</sub> population exposure without share attributable to road traffic<sup>109</sup>



<sup>108</sup> Filliger P., Puybonnieux-Textier V., Schneider J. (1999), Health Costs due to Road Traffic-related Air Pollution, PM<sub>10</sub> Population Exposure, p. 10.

<sup>109</sup> Filliger P., Puybonnieux-Textier V., Schneider J. (1999), Health Costs due to Road Traffic-related Air Pollution, PM<sub>10</sub> Population Exposure, p. 10.

**The population exposure without PM<sub>10</sub> fraction attributable to road traffic** is shown in Figure 6. Compared to total PM<sub>10</sub>, the frequency distribution changes considerably. Most people would live in areas with PM<sub>10</sub> values less than 20 µg/m<sup>3</sup>. In France and Switzerland, less than 3% of the population would live in areas with PM<sub>10</sub> greater than 20 µg/m<sup>3</sup>. In Austria, this portion is higher due to an increased non-traffic caused regional PM<sub>10</sub> background. However, in all three countries, the reduction of the percent values in higher PM<sub>10</sub> concentration classes is substantial and indicates that road traffic contributes considerably to these PM<sub>10</sub> concentration classes.

**Population weighted PM<sub>10</sub> averages** are summarised in Table 7. Interpreting the figures one has to be aware of the fact that PM<sub>10</sub> due to road traffic varies considerably spatially. In city centres, the relative contribution of road traffic to total PM<sub>10</sub> is higher than in rural areas. Typical values, derived from the Swiss model are: 40 - 60% in cities and < 30% in rural areas.

Table 7. **Population weighted annual PM<sub>10</sub> averages for the three countries (calculated from the original grid values of the PM<sub>10</sub> maps)<sup>110</sup>**

	PM10 concentration in µg/m <sup>3</sup> (annual mean)		
	Austria	France	Switzerland
Total PM <sub>10</sub>	26.0	23.5	21.4
PM10 without fraction attributable to road traffic	18.0	14.6	14.0
PM10 due to road traffic	8.0	8.9	7.4

Despite the different methods used, the **results of the three countries are similar**, especially concerning PM<sub>10</sub> levels caused by road traffic. The differences in total PM<sub>10</sub> can be explained by the fact that (a) the background concentration is higher in Central and Eastern Europe than in the Western parts of Europe and (b) for Switzerland, large areas at higher altitudes have significantly lower PM<sub>10</sub> levels. Furthermore, the sulphate fraction of the background concentration may increase from Western to Eastern Europe, resulting in an increase of the non-traffic related PM<sub>10</sub> fraction. However, further investigations including measurements of PM<sub>10</sub> as well as PM<sub>10</sub> components are needed to explore in detail the significance of the differences found.

#### 4. The monetary valuation of air pollution related health effects

Monetarising health effects or even fatalities is often criticised outside the community of economic science. In the general public's opinion it is argued, that human life cannot be expressed in monetary terms. This criticism is based on a misunderstanding as the economic science does not try to assess the value of a specific life. What is being measured is the benefit of a risk reduction due to a lower level of air pollution leading to a decrease in frequency of the different health outcomes.

<sup>110</sup> Filliger P., Puybonnieux-Textier V., Schneider J. (1999) Health Costs due to Road Traffic-related Air Pollution, PM<sub>10</sub> Population Exposure, p. 11.

For this type of assessment, the term „value of preventing a statistical fatality” (VPF) is often used in economic theory. It reflects the fact that a decrease in risk is valued before the negative results have already taken place. Hence, it does not value „ex post” a specific human being’s life lost due to an air pollution related disease.

#### 4.1 *Monetary Evaluation of Mortality*

There are two main different approaches to assess the monetary value of mortality<sup>111</sup>:

- **The gross production / consumption loss:** The costs of additional mortality cases are assessed according to the loss in income / production or the loss of consumption. This valuation concept - sometimes referred as discounted future earnings - is based on the loss resulting from a premature death for the economy as a whole. It is a concept based on the general society, without regarding the individual difference in valuing lower or higher risks of mortality or fatal accidents. The measurement is limited to material aspects of life only, it neglects the intangible costs such as pain, grief and suffering of the victims and their relatives. The main advantage of this approach lies in its simple and transparent calculation concept. Therefore it may be a suitable input for political discussions on policy measures for a reduction of air pollution or other environmental impacts.

However, the main disadvantages are the following:

- The individual aversion against premature death is not considered in this approach, since it only covers material consequences of a fatality.
- Based on the loss for the society as a whole, the concept is in conflict with a basic principle of (welfare-) economic theory according to which each valuation has to be based on the variations in the utility of the concerned individuals.
- An appropriate discount rate has to be chosen which has major implications for the valuation.

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<sup>111</sup> For a detailed discussion see: Sommer H., Seethaler R., Chanel O., Herry M., Masson S., Vergnaud J.-Ch. (1999), Health Costs due to Road Traffic-related Air Pollution, p. 22-26.



- **Willingness to pay (WTP) / Value of preventing a statistical fatality (VPF):** This approach attempts to estimate the demand (the willingness-to-pay) for an improved environmental quality. The central question is, how much individuals are ready to pay to improve their own security or the security of other people. Thus, the sum of individual willingness-to-pay indicates how much value is attributed to an improvement in security or a reduction of environmental impact by the society as a whole. The valuation of a risk reduction in mortality or the value of preventing a “statistical” fatality is calculated by dividing the individual willingness-to-pay values for a risk reduction by the observed change in risk.<sup>112</sup>

The main advantage of the willingness-to-pay approach lies in evaluating the individual preferences for risk reductions of morbidity and premature fatalities. It therefore meets the requirements of welfare economics, since it reflects the individual point of view.

However, a number of arguments against this method are often raised:

- The willingness-to-pay approach depends on the level of income which may pose ethical problems when applied to very different countries (OECD vs. less developed countries).
- If part of income losses are borne by the social insurance system of the country, this loss will not be considered by the individual, although it is part of the society’s costs.
- It is often difficult for the individual to be sufficiently aware of the risk level at stake and the consequences on health. Individuals may not be familiar with small variations in risk which may imply large discrepancies between individual valuations.
- The main difficulty of the WTP approach lies in obtaining reliable and correct empirical estimations, because results are highly sensitive to the survey design.

Nevertheless, recent research provides promising results. The chosen WTP values for the present study are based on a contingent valuation method, in which the direct comparison between money and risk of mortality is replaced by a sequence of chained interviews.<sup>113</sup>

Based on this discussion the Willingness-to-pay (WTP) for the Value of a Prevented Fatality (VPF) was used as common methodological approach.<sup>114</sup>

Unfortunately, so far no empirical studies have been carried out specifically for air pollution related mortality risk. Furthermore, under the prevailing budget and time constraint it was out of scope to conduct an empirical survey within this project. Therefore, empirical results of road accident related WTP were used as a starting point.

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<sup>112</sup> Example: A policy measure is able to reduce the yearly risk of fatal road accidents from 4 cases per 10’000 to 3 cases per 10’000. For this risk reduction of 1 case per 10’000, the affected individuals are ready to pay an average amount of 100 US \$. In this case, the value of a statistical prevented fatality amounts to 1 million US \$ (100 US \$ /0.0001 risk reduction). Again, it needs to be recognised that the respondents are not asked about their willingness-to-pay for the avoidance of their own death but about the willingness-to-pay for a change in risk.

<sup>113</sup> See Sommer H., Seethaler R., Chanel O., Herry M., Massons S., Vergnaud J.-Ch. (1999), Health Costs due to Road Traffic-related Air Pollution, Annex 7 p. 77-83.

<sup>114</sup> According to the country specific needs, in addition to the WTP-approach an alternative partial assessment approach was conducted, based on the loss of production or consumption (see chapter 5.3).

The most recent studies from the 1990's indicate a range of WTP values for the prevention of a statistical fatality of 0.7 to 6.1 million EUR.<sup>115</sup> The latest empirical study, conducted by Jones-Lee *et al.*<sup>116</sup> provides a VPF of 1.42 million EUR (range: 0.7-2.3 million EUR).

Based on these most recent results and the experience of former studies a **starting value of 1.4 million EUR** is adopted for the value of preventing a statistical fatality. This choice is supported by the use of a similar starting value (1.2 million EUR) in a recent study on behalf of the UK Department for Environment, Transport and Regions (DETR) and the fact that it lies in the lower part of the range of the majority of recent empirical evaluations.<sup>117</sup> This choice is in line with the “at least” approach prevailing throughout the entire project.

Road accident related fatal risk differs from air pollution related risk. The latter is to a large extent involuntary and beyond the responsibility and control of those exposed to it. In addition, while taking the risk of a traffic accident, driving itself offers a direct personal benefit. On the other hand, air pollution related risk is less often connected to a direct personal benefit, although it is to some extent transport induced. Because of this **different risk context**, air pollution related risk aversion is likely to be higher than for fatal road accidents.<sup>118</sup> The impact of the contextual difference between road accident and air pollution related risk on individual aversion is subject of several empirical studies and has produced factors in the range of 1.5 to 2. However, the empirical evidence is not considered to be sufficient and following the “at least” approach, the contextual adaptation of the WTP value is abandoned in the present study.

Based on the available epidemiological literature, a direct conclusion about the age structure of the air pollution related premature deaths is not yet possible. It is, however, known that these fatalities are mostly related to respiratory and cardiovascular diseases and lung cancer. In Austria, France and Switzerland, the average age of these respiratory and cardiovascular fatalities lies between 75 and 85 years (see Figure 8).

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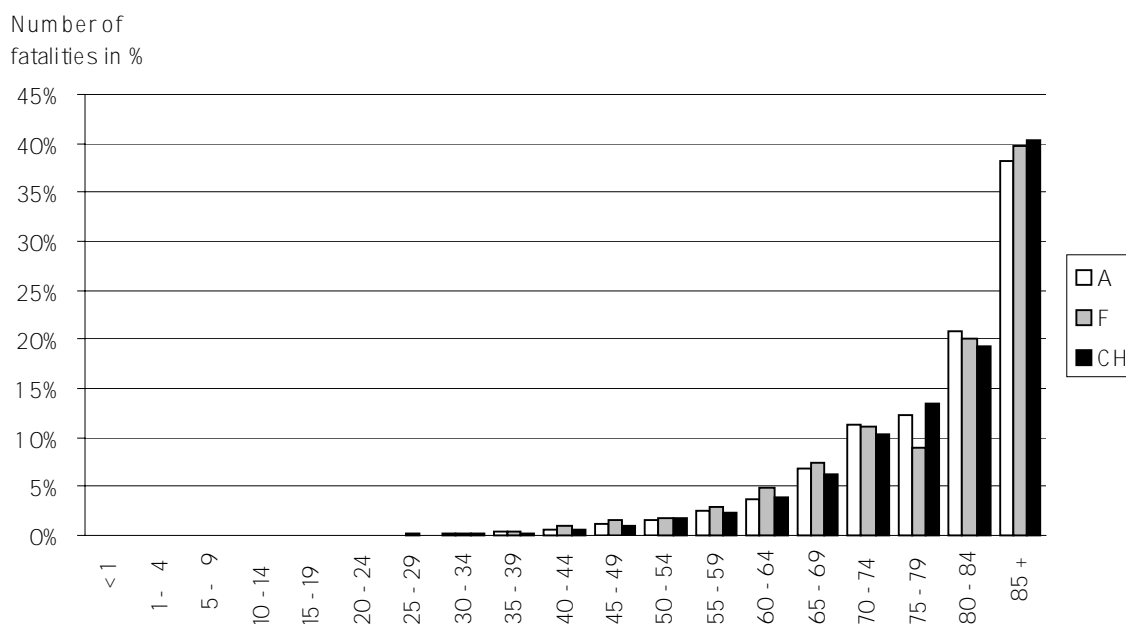
<sup>115</sup> Viscusi W.K. (1993), *The Value of Risks to Life and Health*; Beattie J. et al (1998), *Valuing Health and Safety Controls: A Literature Review*; Institute of Environmental Studies, Norwegian Institute for Air Research, International Institute for Applied System Research (1997), *Economic Evaluation of quality targets for sulphur dioxide, nitrogen dioxide, fine and suspended particulate matter and lead*; ZEW / ISI (1997), *External Quality Evaluation*; ExternE (1995), *Externalities of Energie, Vol 2: Methodology*.

<sup>116</sup> Jones-Lee M. et al (1998), *On the Contingent Valuation of Safety and the Safety of Contingent Valuation: Part 2 - The CV/SG “Chained” Approach*; Chilton S., Covey J., Lorraine H., Jones-Lee M., Loomes G., Pidgeon N., Spencer A. (1998), *New Research Results on the Valuation of Preventing Fatal Road accident Casualties*.

<sup>117</sup> For example, the ExternE-Project, a very extensive project on behalf of the European Community on the external costs of energy use, is based on a meta-analytical value of 2.6 million Euro with a range from 2.1 to 3.0 million Euro. See: ExternE (1995), *Externalities of Energy, Vol. 2: Methodology*.

<sup>118</sup> This view is adopted by a number of authors. See: Jones-Lee et al (1998), *On the Contingent Valuation of Safety and the Safety of Contingent Valuation: Part 1 - Caveat Investigator and Department of Health (1999), Economic Appraisal of the Health Effects of Air Pollution*, p. 63-66.

Figure 8. **Age structure of fatalities due to respiratory, cardiovascular diseases and lung cancer in Austria<sup>119</sup>, France and Switzerland (1996)**



Hence, the average age of the air pollution related fatalities is much higher than for victims of fatal road accidents (30-40 years of age).

Theoretical as well as empirical evidence indicates a decreasing WTP with increasing age, with reduced remaining life expectancy and with reduced quality of life. For the present study, the relationship adopted is provided by the latest research of Jones-Lee.<sup>120</sup> Weighting the age structure of the fatalities due to respiratory and cardiovascular disease and lung cancer in all three countries by the age factor, an average adaptation factor of 61% is obtained for the present willingness-to-pay for a prevented fatality value.

Based on the preceding discussion we used a value of **0.9 Mio. EUR** (=61% x 1.4 Mio. EUR) for the value of preventing a statistical fatality. Hence, the cost reducing adjustment for age is maintained, meanwhile the cost increasing adjustment for the risk context is abandoned. This implies a very strict application of the “**at least**” approach.

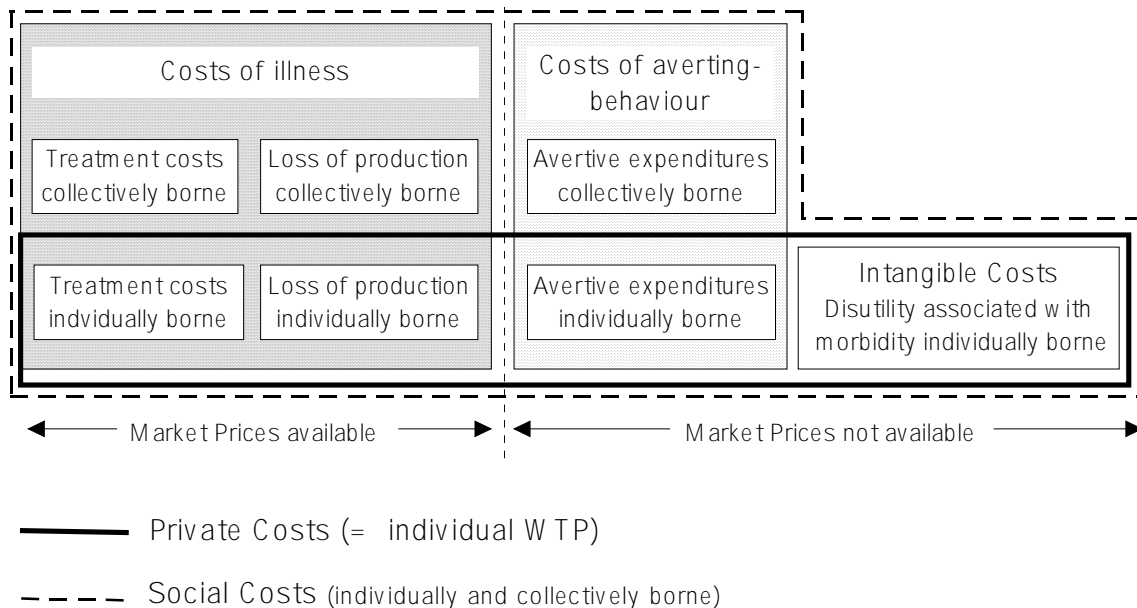
<sup>119</sup> Only respiratory and cardiovascular diseases without lung cancer.

<sup>120</sup> Several studies by Jones-Lee show a reversed U-shaped relationship between the age and the willingness-to-pay. See: Department of Health (1999), Economic Appraisal of the Health Effects of Air Pollution, p. 67 and direct information of M. Jones-Lee (1998).

## 4.2 Monetary valuation of morbidity

From an economic point of view, the costs of morbidity may be subdivided by two main criteria, namely by the cost component and by the entity in charge of paying them. As shown in Figure 9, the costs of illness, the costs of averting behaviour and the intangible costs are three different components. They are either borne privately or in the case of cost of illness and costs of averting behaviour collectively as well.

Figure 9. Overview on the costs of morbidity



**Costs of illness (COI)** contain the loss of production due to a possible incapacity to work and the medical treatment costs. They determine the “material part” of the health costs and may be assessed on the basis of real market prices (loss of earnings, costs for medicaments, costs per day in hospital, etc.).

**Costs of averting-behaviour** result from changes in behaviour due to air pollution. The abstention from outdoor sport activities during a summer day with high ozone concentration, the installation of air filters or a different choice of residential location to avoid high levels of air pollution are some current examples. The higher the costs of avoidance measures, the smaller will probably be the number of air pollution related morbidity cases. Considering the extent of avoidance measures taken so far, neglecting the costs of averting-behaviour may result in a considerable underestimation of the morbidity costs. However, for the assessment of these costs market prices are mostly non-existent.

The third essential component of morbidity costs are the **intangible costs** reflecting the individual loss of the victims utility and consisting of pain, grief and suffering due to a disease. Based on empirical evidence, the risk aversion of morbidity is mainly determined by these inconveniences (losses in utility).

In order to draw a complete picture of the total morbidity costs, individually borne private costs and the costs borne collectively, e.g. by a social security system, have to be considered. All components together constitute the social costs of morbidity.

Similar to the methodological possibilities for the monetary valuation of mortality, the morbidity may be assessed with different methodological approaches. For the costs of illness (COI) containing the production loss and medical treatment costs, the damage cost approach is used. Based on market prices, it assesses all individually as well as collectively borne material costs. However, for the costs of averting-behaviour and the intangible costs, this approach is not suitable, since market prices are mostly non-existent.

The willingness-to-pay approach focuses on the individually borne costs (private costs). It establishes the individuals utility of a risk reduction in air pollution related morbidity and reflects all costs the individual expects to bear in case of a disease, such as loss of earnings, costs of averting behaviour or intangible costs.

As mentioned above, the advantage of the willingness-to-pay approach consists of its integration of material and intangible costs, that cannot be measured by any other method but are often considerably higher than material costs. However, the disadvantage is its limitation to individually borne costs, especially when a large part of health costs is borne by collective means.

In spite of this limitation the willingness-to-pay approach is considered to be a better approximation of social costs of morbidity than the COI approach. Therefore we used the WTP-approach as the main common methodological framework to assess the morbidity costs.

Unfortunately, the literature on WTP based, air pollution related, morbidity costs is very rare in Europe and most available studies refer to the US context. Their application to Europe is not completely unproblematic, since a recent research study provide lower results for a European country.<sup>121</sup> The different socio-cultural background and the difference in health care and insurance systems ask for an application of country specific WTP results. In spite of this problem, the present study had to be based on existing values since the available resources did not allow for an empirical survey within this project.

Table 10 presents the WTP for avoiding different air pollution related health outcomes.

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<sup>121</sup> Navrud S. (1998), Valuing Health Impacts from Air Pollution in Europe - New Empirical Evidence on Morbidity.

Table 10. **TP for the avoidance of air pollution related health outcomes**

Health indicator	WTP-Value (EUR)	
Respiratory Hospital Admission	7'870	per admission <sup>122</sup>
Cardiovascular Hospital Admission	7'870	per admission <sup>23</sup>
Chronic Bronchitis	209'000	per case <sup>123</sup>
Bronchitis	131	per case <sup>124</sup>
Restricted Activity Day	94	per day <sup>25</sup>
Asthmatics: Asthma Attacks (person day)	31	per attack <sup>125</sup>

## 5. Results

### 5.1 Quantitative results of $PM_{10}$ related health effects

From the epidemiological data (fixed base line increment per  $10 \mu\text{g}/\text{m}^3$   $PM_{10}$  per 1 million inhabitant) on the one hand and the average exposure level of the population on the other hand, the number of health outcomes can be determined.

These calculations may be done for the current exposure to particulate matter as well as for a hypothetical situation **without** road traffic-related air pollution. The difference between the two results corresponds to the number of morbidity and mortality cases attributable to road traffic-related air pollution.

In Table 11 for Austria, France and Switzerland, the health effects considered are presented for the average annual exposure to total air pollution and for the average annual exposure to road traffic-related air pollution. According to the epidemiological foundations, for each health outcome the respective age group is considered. Knowing the distribution of the different population groups across exposure classes (chapter 3) and the parameters of the exposure-response function (chapter 2), the absolute number of health outcomes may be established for each country with or without the road traffic-related share of air pollution.

<sup>122</sup> Based on ExternE (1995), Externalities of Energy, Volume 2, Methodology, Part II: Economic Valuation, p. 519, adjusted for inflation according to Nilsson M., Gullberg M. (1998), Externalities of Energy, Swedish Implementation of the ExternE Methodology.

<sup>123</sup> Chestnut L.G. (1995), Human health benefits from sulfate reductions under Title IV of the 1990 clean air act amendments, p. 5-20, WTP for an average chronic bronchitis case.

<sup>124</sup> Maddison D. (1997), Valuing the morbidity effects of air pollution, p. 8.

<sup>125</sup> Ostro B., Chestnut L. (1997), Assessing the Health Benefits of Reducing Particulate Matter Air Pollution in The United States, p. 100.

It needs to be emphasized that the health effects are **only considered from the exposure class of 5-10  $\mu\text{g}/\text{m}^3$   $\text{PM}_{10}$  onwards** (average 7.5  $\mu\text{g}/\text{m}^3$   $\text{PM}_{10}$ ). This restriction reflects the fact that epidemiological studies have not yet included the exposure-response relationship below this level. In addition, it needs to be considered that there is a natural background concentration level which is not man made. For Austria, France and Switzerland this natural baseline pollutant level is estimated to be <7.5  $\mu\text{g}/\text{m}^3$   $\text{PM}_{10}$ . For the further assessment of air pollution measures it is adequate to only consider the air pollution of human activities.

In Table 11, the negative effects of air pollution are divided into the number of health outcomes related to total air pollution and those related to the road traffic share only.

### 5.1.1 *Mortality*

In 1996, air pollution caused **5'600 cases of premature death** in Austria, **31'700 cases** in France and **3'300 cases** in Switzerland. In Austria 2'400, in France 17'600 and in Switzerland 1'800 cases are attributable to road traffic-related air pollution.

According to the epidemiological foundations, the increase in premature mortality is only considered for adults  $\geq 30$  years of age and for the exposure class of 5-10  $\mu\text{g}/\text{m}^3$   $\text{PM}_{10}$  (class mean 7.5  $\mu\text{g}/\text{m}^3$  ) onwards.

**Table 11. Additional cases of mortality and morbidity due to air pollution in Austria, France and Switzerland<sup>126</sup>**

Health outcome	Cases or days attributable to total air pollution			Cases or days attributable to road traffic		
	Austria	France	Switzerland	Austria	France	Switzerland
Long-term mortality (adults ≥ 30 years; excluding violent death)	5'600 3'400 - 7'800	31'700 19'200 - 44'400	3'300 2'000 - 4'700	2'400 1'500 - 3'400	17'600 10'700 - 24'700	1'800 1'100 - 2'500
Respiratory hospital admissions (all ages)	3'400 400 - 6'500	13'800 1'400 - 26'300	1'300 140 - 2'500	1'500 160 - 2'800	7'700 800 - 14'600	700 70 - 1'300
Cardiovascular hospital admissions (all ages)	6'700 3'500 - 10'000	19'800 10'400 - 29'400	3'000 1'500 - 4'400	2'900 1'500 - 4'300	11'000 5'800 - 16'300	1'600 800 - 2'400
Chronic bronchitis incidence (adults ≥ 25 years)	6'200 600 - 12'000	36'700 3'300 - 73'100	4'200 370 - 8'400	2'700 240 - 5'300	20'400 1'800 - 40'700	2'300 200 - 4'500
Bronchitis (children < 15 years)	48'000 21'000 - 86'000	450'000 198'500 - 813'600	45'000 20'000 - 82'000	21'000 9'000 - 37'000	250'000 110'000 - 453'000	24'000 11'000 - 44'000
Restricted activity days (adults ≥ 20 years)	3'100'000 2'600'000 - 3'600'000	24'600'000 20'700'000 - 28'500'000	2'800'000 2'400'000 - 3'200'000	1'300'000 1'100'000 - 1'600'000	13'700'000 11'500'000 - 15'900'000	1'500'000 1'200'000 - 1'700'000
Asthmatics: asthma attacks (children < 15 years, person days)	35'000 21'000 - 48'000	243'000 149'000 - 337'000	24'000 15'000 - 33'000	15'000 9'000 - 21'000	135'000 83'000 - 188'000	13'000 8'000 - 17'000
Asthmatics: asthma attacks (adults ≥ 15 years, person days)	94'000 46'000 - 143'000	577'000 281'000 - 879'000	63'000 30'000 - 95'000	40'000 20'000 - 62'000	321'000 155'000 - 489'000	33'000 16'000 - 51'000

126

Table printed with permission from Lancel, Kinzli N, et al (2000), Public Health Impact of Outdoor and Traffic-related Air Pollution: A Tri-national European Assessment, in press.



### 5.1.2 *Morbidity*

Within the **additional morbidity cases**, the highest incidence in all three countries is registered for **acute bronchitis in children** younger than 15 years. Some 21'000 cases in Austria, some 250'000 cases in France and some 24'000 cases in Switzerland were attributable to road traffic-related air pollution in 1996.

The second highest frequency is obtained for the incidence of **chronic bronchitis in adults**. In 1996, the number attributable to road traffic-related air pollution amounts to ca 2'700 cases in Austria, 20'400 cases in France and 2'200 cases in Switzerland.

The additional cases of **cardiovascular hospital admissions** (all ages) due to road traffic-related air pollution amount to 2'900 cases in Austria, 11'000 cases in France and 1'600 cases in Switzerland. The smallest number of road traffic attributable cases is obtained for **respiratory hospital admissions** (all ages). In 1996, it amounts to ca 1'500 cases in Austria, 7'700 cases in France and 700 cases in Switzerland.

Concerning the **additional days** of air pollution related morbidity, a very large number of **restricted activity days** for adults ( $\geq 20$  years) results in all three countries. In 1996, in Austria, 1.3 million days, in France 13.7 million days and in Switzerland 1.5 million days with restricted activity are attributed due to road-traffic-related air pollution.

In 1996, for Austria 15'000 **asthma attacks in children** ( $<15$  years) and **40'000 asthma attacks in adults** ( $\geq 15$  years) are attributable to road traffic-related air pollution. France and Switzerland attributed 135'000 and 13'000 asthma attacks in children and 321'000 and 33'000 asthma attacks in adults to road traffic-related air pollution.

### 5.2 *Health costs due to air pollution based on the willingness-to-pay approach*

Based on the **willingness-to-pay** approach, in 1996 the total air pollution in Austria, France and Switzerland caused a high level of health costs. The **total air pollution related health costs** across the three countries amount to **49'700 million EUR** (Table 12), of which **26'700 million EUR** are attributable to **road traffic-related air pollution**.

In Austria (6'700 million EUR) and Switzerland (4'200 million EUR) the **total air pollution** related health costs reach a similar level. Due to the much larger population, the French costs amount to 38'800 million EUR.

**Table 12. Health costs due to road traffic-related air pollution in Austria, France and Switzerland based on the willingness-to-pay approach (1996)**

	Austria		France		Switzerland	
	Total costs with road traffic share	Costs attributable to road	Total costs with road traffic share	Costs attributable to road	Total costs with road traffic share	Costs attributable to road
Costs of mortality (million EUR)	5'000 3'000 - 7'000	2'200 1'300 - 3'000	28'500 17'300 - 39'900	15'900 9'600 - 22'200	3'000 1'800 - 4'200	1'600 1'000 - 2'200
Costs of morbidity (million EUR)	1'700 400 - 3'000	700 200 - 1'300	10'300 2'800 - 18'500	5'700 1'500 - 10'300	1'200 300 - 2'100	600 200 - 1'100
Total costs (million EUR)	6'700 3'400 - 10'000	2'900 1'500 - 4'300	38'800 20'100 - 58'400	21'600 11'100 - 32'500	4'200 2'100 - 6'300	2'200 1'200 - 3'300

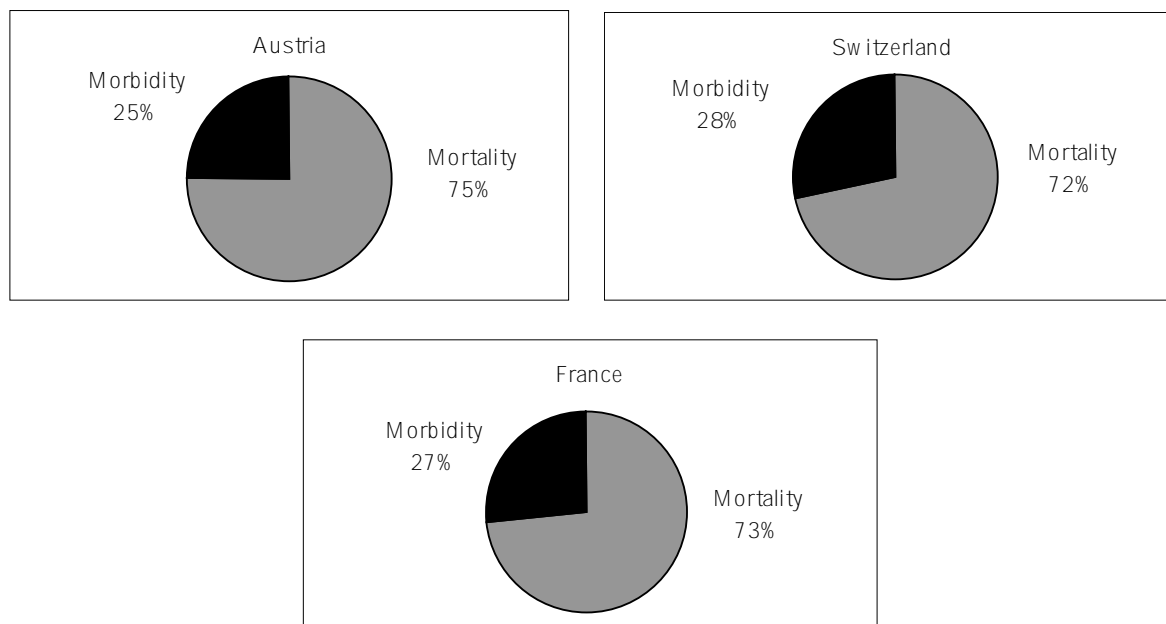
	all three countries	
	Total costs with road traffic share	Costs attributable to road
Costs of mortality (million EUR)	36'500 22'100 - 51'100	19'700 11'900 - 27'400
Costs of morbidity (million EUR)	13'200 3'500 - 23'600	7'000 1'900 - 12'700
Total costs (million EUR)	49'700 25'600 - 74'700	26'700 13'800 - 40'100

In all three countries, road traffic is a main source of air pollution related health costs. The absolute level of the road traffic-related air pollution amounts to 8.9  $\mu\text{g}/\text{m}^3$   $\text{PM}_{10}$  in France, 8.0  $\mu\text{g}/\text{m}^3$  in Austria and of 7.4  $\mu\text{g}/\text{m}^3$  in Switzerland (as population weighted annual averages). It needs to be remembered that tailpipe exhaust is only responsible for part of the  $\text{PM}_{10}$  concentration. The considerable proportion of other emissions, such as tyre wear, other abrasion products and road dust re-suspension are independent of the share of diesel engines.

The lower relative proportion of traffic-related health costs in Austria may be caused by a higher background of  $\text{PM}_{10}$  in 1996 which may contain a high sulphate amount (especially in Eastern Austria).

Depending on the country, 72% to 75% of the health costs are related to mortality (see Figure 13). The differences are mainly due to country specific differences in the baseline frequencies of the health outcomes observed.

Figure 13. **Breakdown of air pollution related costs by mortality and morbidity**

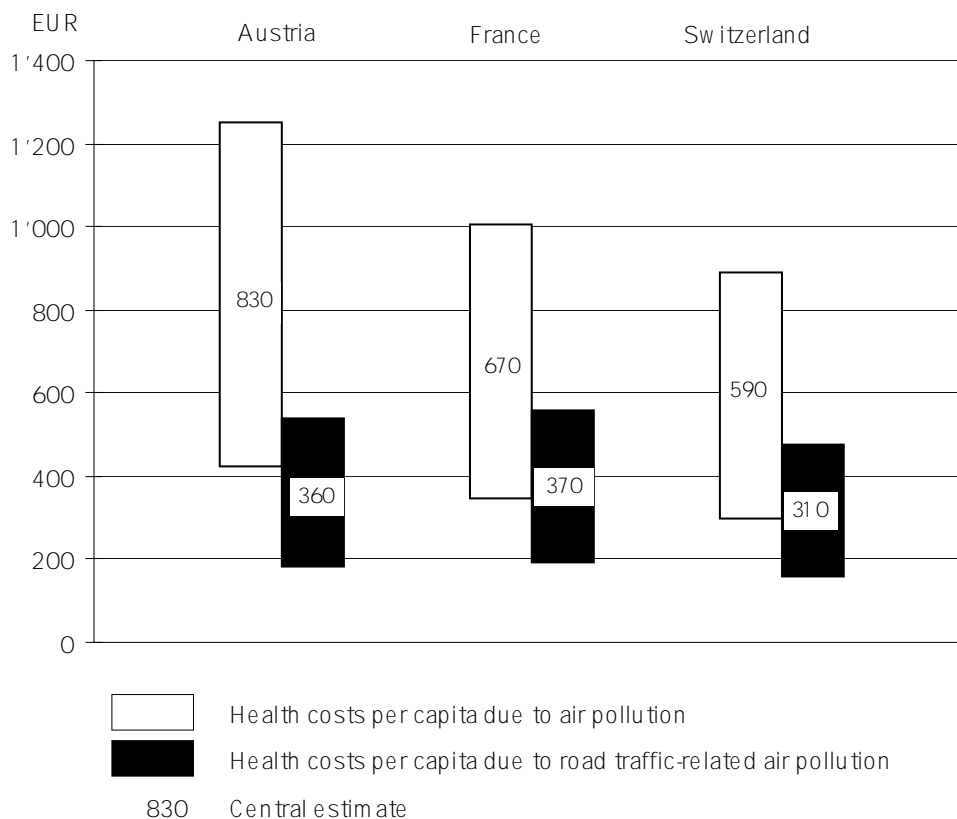


Comparing the **total air pollution related health costs per capita** (see Figure 14) the results of the three countries stay within the same range, **although** the central estimates indicate differences between the three countries. The highest per capita costs are shown for Austria.

For the **road traffic-related health costs**, the per capita results differ much less between the three countries: The highest value is obtained in France with about 370 EUR per capita, followed by Austria with about 360 EUR per capita and Switzerland with about 310 EUR per capita.

These differences are mainly due to air pollution levels (average level of population weighted total  $PM_{10}$  exposure and the traffic-related share) and the epidemiological results (different national mortality and morbidity rates in general). However, the results of the three countries stay within the same range. Therefore, the differences in per capita costs mentioned above should not be overinterpreted.

Figure 14. **Air pollution related health costs per capita based on the willingness-to-pay approach (1996)**



### 5.3 *Partial assessment approach: health costs due to air pollution based on gross production loss approach / cost of illness (COI)*

According to the country specific needs, in addition to the WTP-approach a partial assessment approach has been used to evaluate the health costs:

- The mortality related health costs are based on the production/consumption loss. The losses are determined on the potential years of life lost.
- The morbidity related health costs are based on the costs of illness, which consist of the production loss due to a incapacity of work and the medical treatment costs.

The partial assessment approach is an extreme implementation of the “at least” approach in so far, as it does not include a major aspect of mortality and morbidity risk related costs, namely the intangible costs. In addition, for some health outcomes (chronic bronchitis, asthma attacks) only the medical treatment costs are included, as for the production loss related to these health outcomes, no data is presently available. In absence of empirical data, for the very great number of restricted activity days no costs of production loss and medical treatment could be established at all.

The per capita costs of the partial assessment approach are shown in Table 15.

**Table 15. Air pollution related health costs per capita based on the partial assessment approach (1996)**

	Austria		France		Switzerland	
	Total costs with road traffic share	Costs attributable to road traffic	Total costs with road traffic share	Costs attributable to road traffic	Total costs with road traffic share	Costs attributable to road traffic
Costs per capita (EUR)	1 40 80 - 190	60 30 - 80	70 40 - 100	40 20 - 60	1 60 1 00 - 230	90 50 - 120

All the above mentioned restrictions for the assessment of health costs due to air pollution reduces the costs by a factor of 3.6 (in Switzerland) up to a factor of 9.1 (in France) compare to the willingness to pay based results.

The differences between the countries are mainly based on the country specific calculation methods. Different cost levels for the production or consumption loss approach have been used: 18'230 EUR per year of life lost in Austria, 12'600 in France and 34'800 in Switzerland.<sup>127</sup> The use of the same valuation per year of life lost for the three countries would have suppressed most of the differences in relative ratios between WTP and partial assessment results.

#### **5.4 Interpretation and sensitivity of the results**

For the assessment of air pollution related health costs, different methodological approaches are available. For an integral view, considering the material and intangible costs, the willingness-to-pay approach for the monetary valuation of mortality and morbidity costs comes to the fore.

Based on this approach, the results may be interpreted as follows:

- In all three countries, road traffic is a main source of air pollution related health costs. The absolute level of the road traffic-related costs stay within the same range: 0.9%-2.7% of the GDP in France, 0.8%-2.5% in Austria and 0.6%-1.7% in Switzerland.
- Compared to other road traffic-related negative impacts (noise, accidents, damage to buildings), the health costs are considerable. According to comparative studies in Austria and Switzerland, the health costs exceed the present estimations of accident costs.

<sup>127</sup>

The Swiss value contains an amount of 14'200 EUR per year of life lost as a low and insufficient proxy for the intangible costs. The proxy is based on compensation payments granted by courts.

- Based on the actual air pollution, a reduction in the average PM<sub>10</sub> exposure of 10 µg/m<sup>3</sup> would result in the long run in a annual cost reduction of 3'600 million EUR in Austria, 24'300 million EUR in France and 3'000 million EUR in Switzerland. However, it needs to be borne in mind that the health costs (assessed by the willingness-to-pay approach) are mostly borne by individuals through welfare losses and intangible costs. Therefore, the cost savings due to a reduction of air pollution don't result in a similar reduction of the health budget covered by the social insurance system.
- The cost reduction has to be seen as a long-term effect and that the savings during transition years would be less.

The sensitivity of the overall results is influenced by all three partial steps (the assessment of exposure, the exposure-response relationship for mortality and morbidity, the monetary valuation of mortality and morbidity related risk). The impact of key assumptions and methodological decisions has been quantified in the health impact paper<sup>128</sup>, and discussed in more detail in our full reports.<sup>129</sup> In general, for each sensitive assumption an “at least” approach was adopted. The real costs of (road traffic-related) air pollution are considered to be higher than the results of the present study, since:

- various PM<sub>10</sub> related health effects (e.g. infant mortality) were not considered due to the absence of available data;
- the additional effects of other pollutants (e.g. ozone) were not considered;
- for the monetary valuation generally lower estimates of cost factors were chosen.

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<sup>128</sup> Künzli N. et al (2000), Public Health Impact of Outdoor and Traffic-related Air Pollution: A Tri-national European Assessment, in Lancet, in press.

<sup>129</sup> Filliger P., Puybonnieux-Textier V., Schneider J. (1999), Health Costs due to Road Traffic-related Air Pollution, PM<sub>10</sub> Population Exposure; Künzli N., Kaiser R., Medina S., Studnicka M., Oberfeld G., Horak F. (1999); Health Costs due to Road Traffic-related Air Pollution, Air Pollution Attributable Cases; Sommer H., Seethaler R., Chanel O., Herry M., Massons S., Vergnaud J.-Ch. (1999), Health Costs due to Road Traffic-related Air Pollution; see also Künzli N. et al (2000), Public Health Impact of Outdoor and Traffic-related Air Pollution: A Tri-national European Assessment, in press.

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# ESTIMATING THE ANCILLARY BENEFITS OF GREENHOUSE GAS MITIGATION POLICIES IN THE US

by Dallas BURTRAW and Michael A. TOMAN

## 1. Introduction

To a large extent, policies for limiting emissions of greenhouse gases (GHGs) have been analyzed in terms of their costs and potential for reducing the rate of increase in atmospheric concentrations of these gases. However, actions to slow atmospheric GHG accumulation could have a number of other impacts, such as a reduction in conventional environmental pollutants. The benefits (or costs) that result are often referred to as “ancillary” to the benefits and costs of GHG abatement (though there is controversy surrounding this terminology and the underlying concepts).

A failure to adequately consider ancillary benefits and costs of GHG policy could lead to an inaccurate assessment of the overall impacts of mitigation policies. In particular, not accounting for ancillary benefits and costs would lead to an incorrect identification of a “no regrets” level of GHG mitigation. It also could lead to the choice of an unnecessarily expensive policy because of its failure to fully exploit potential ancillary benefits.

In this paper we first review briefly the concept of ancillary benefit, as it is developed in more detail in other papers in this proceedings and elsewhere in the literature. The concept turns out to be surprisingly difficult to define precisely. What is considered an ancillary benefit depends on the scope of policies being considered, the policy objectives being pursued, and the identity of the interests being served. That said, however, we describe what we believe is a serviceable definition of ancillary benefits from the perspective of evaluating GHG mitigation policies within the “Annex I” countries who would have emission limitation obligations under the Kyoto Protocol. We focus on mitigation in this paper, while acknowledging that adaptation policies also could have ancillary effects (for example, improved surveillance of tropical diseases could yield immediate health dividends; protection of coastal lands could harm wetland habitats in the more immediate term).

Having established a workable definition, we then turn to issues related to measuring ancillary benefits. To illustrate these issues, we consider how lower GHG levels resulting from less fossil fuel use could also reduce various “criteria” air pollutants (as defined in the U.S. Clean Air Act). Recent comprehensive studies of electricity fuel cycles indicate that the lion’s share of the environmental and public health effects of fuel and technology choices in electricity generation stem from air emissions. These effects typically total about 85 percent or more of the quantifiable environmental concerns, excluding climate change and species biodiversity (Lee *et al.*, 1995; Rowe *et al.*, 1995; EC, 1995). Thus, focusing on the air-health pathway of ancillary benefits is likely to provide a fairly reliable picture of total ancillary benefits, though controversy remains regarding the magnitudes of non-health effects.

We find in our review that estimates of ancillary health benefits for the US vary considerably. Previous estimates have ranged from over \$60 per ton of carbon reduced (or greater in one special context) to \$3 per ton. The dispersion of US estimates reflects partly underlying parametric uncertainty, in particular the economic valuation of health impacts. There also are several important differences in the identification of baselines, in particular the effect of current and future regulatory standards for conventional pollutants (see the paper by Morgenstern). Still other differences arise in the scale of modeling, in particular the distinction between more detailed sector-specific analyses (in practice these involve the electricity sector), and specific geographic locations, in contrast to economy-wide estimates based on much simpler modeling of environmental impacts.

For a variety of reasons that are evident below, we have much more confidence in more conservative (lower) estimates of ancillary benefits (especially those drawn from more detailed models) compared to estimates that equal or exceed the costs of GHG control. Ancillary benefits could offset a significant fraction of the costs of carbon reduction with moderate GHG policies. It also may be possible to orient GHG abatement policies in certain ways to take greater advantage of ancillary benefits. However, the considerable variation in baseline assumptions and in policy scenarios, coupled with uncertainty about the size of ancillary benefits, leads to tremendous variation in estimates, precluding identification of a single “best estimate” of their magnitude.

In the next section of the paper we briefly review our working definition of ancillary benefit. Section III contains a review of estimates from a number of US studies. Section IV provides a more detailed discussion of methods and results from ongoing research at RFF on ancillary benefits from GHG restrictions in the electricity sector. This approach reflects what might be seen as a “best practice” in the development and use of detailed methodology for linking GHG policies to changes in conventional pollutant emissions, ambient consequences, health effects assessment, and economic valuation. Section 5 interprets and critiques the various estimates presented in Sections 3 and 4. Section 6 offers concluding remarks and suggestions for further research.

## **2. Defining and measuring ancillary benefit or cost**

### **2.1 *Definitional issues***

An ancillary benefit of a GHG mitigation policy is understood by many analysts to refer to a benefit (derived from GHG mitigation) that is reaped in addition to the benefit targeted by the policy, which is reduction in the adverse impacts of global climate change. An ancillary cost would be a negative impact experienced in addition to the targeted benefit. The key elements of this definition, and the sources of much of the controversy surrounding the notion of ancillary, are “in addition” and “targeted.”

In the context we have used for defining ancillary benefits and costs, the principal policy goal is GHG mitigation in order to reduce adverse climate impacts. Asserting that ancillary benefits are additional to the benefits of reducing climate change does not mean these benefits are necessarily less important, or that other policy goals are less important than addressing climate change. Benefits that are ancillary to climate change could be bigger in magnitude and more salient for the affected citizens and their decision makers. Our definition simply puts ancillary benefits in a certain policy context.

That policy context can be and is debated. Developing countries have argued with justification that they have more pressing development and environmental needs than reducing their GHGs. In this broader policy context, what we refer to as ancillary benefits could be considered as “co-benefits” of policies designed to promote various objectives. Our own view is that when discussing climate change policies, the benefits and costs targeted by the policies should be considered as those associated with GHG mitigation and climate change risk reduction; other benefits and costs should be treated as ancillary in the sense we have defined the term above, but not given short shrift.

Some more specific but related considerations that arise in defining ancillary benefits and costs involve the scope of what is included in the calculation and the perspective of the decision maker evaluating benefits and costs. A number of kinds of impacts can be considered when evaluating ancillary benefits and costs. Much of the emphasis in these calculations has been on near-term health impacts in relatively close proximity to the GHG mitigation (for example, reduced incidence of lung disease in the same area as a coal-fired power plant if that plant is used less as a consequence of GHG mitigation measures), but a variety of other impacts also could be important.

For example, ecological systems could be affected by reductions in the flow of conventional pollutants (for example, less fossil fuel use could mean less nitrogen oxide deposition into water bodies). Reduced pollutants also could reduce some direct costs, such as maintenance of infrastructure and pollution-related reductions in crop yields. Also, traffic accidents could be reduced from less driving or slower traffic speeds. Reduced traffic could lower road maintenance costs. Similarly, increased forest areas dedicated to carbon sequestration could increase recreational opportunities and reduce erosion. GHG policies could also stimulate technical innovation.

Ancillary costs can arise if energy substitution leads to other health and environmental risks (e.g. from nuclear power, uncontrolled particulate emissions from biomass combustion, or use of diesel fuel in lieu of gasoline, since diesel fuel has lower carbon emissions but greater emissions of other pollutants). Better building insulation can add to indoor air pollution, including radon, and switching from coal to gas raises the specter of fugitive emissions of methane, a more potent greenhouse gas than CO<sub>2</sub>. Also, policies that promote reforestation could encourage destruction of old growth natural forests because younger forests allow more carbon storage. Further, GHG mitigation policies could mainly redirect innovation efforts away from other productive activities, rather than increasing it. In addition, relatively expensive GHG mitigation policies could have some negative side effects on health by reducing the resources available to households for other health-improving investments.

An economic perspective on ancillary benefits sees them as part of a larger concern with economic efficiency, as typically expressed in measures of aggregate benefits and costs. From this perspective, it is important not to isolate ancillary benefit and cost information from other relevant benefit and cost information associated with GHG policy. Ancillary benefits of a policy could be substantial, but they are nonetheless a questionable achievement if the cost of garnering these benefits is much larger. Often ancillary benefits are expressed in terms of a monetary measure per ton of carbon not emitted to the atmosphere as a consequence of the mitigation policy. Expressed this way, ancillary benefits (and costs) can be compared to the cost of mitigation. This is usually a meaningful and useful comparison, since ancillary benefits often (but not always) occur on the same relatively shorter-term time scale as mitigation costs, while the benefits of reducing climate change will be realized in the more distant future.

A final related point is that the scope and magnitude of ancillary benefits and costs depends on the perspective of the decision maker as to what constitutes policy relevant impacts. From the perspective of a hypothetical global decision maker concerned with global social well-being, ancillary benefits and costs are important wherever they are incurred. From this perspective it thus is important to consider how a redistribution in the location of GHG mitigation could affect ancillary benefits and costs.

In particular, policy mechanisms like international emissions trading or the Clean Development Mechanism will redistribute ancillary impacts toward those countries undertaking more GHG mitigation. And efforts by Annex I alone to mitigate GHGs could have collateral effects in developing countries not bound by quantitative emissions limits, in that lower energy prices in international markets will stimulate some additional energy use and associated local environmental effects in those countries. On the other hand, for an Annex I decision maker evaluating the benefits and costs of GHG mitigation policies in his or her own jurisdiction, the relevant ancillary benefits and costs are likely to consist primarily of those affecting individuals in that political jurisdiction. Cross-boundary spillovers like those illustrated above are relevant for the Annex I decision maker only to the extent that a sense of ethical responsibility or altruism motivates a broader concern for the spillovers.

Still another perspective would be adopted by the developing country decision maker contemplating involvement in the Clean Development Mechanism. In this case, the primary benefits in terms of importance for the developing country considering hosting a GHG-reducing investment are likely to be the benefits that are ancillary to the GHG control according to our definition of the term.

## **2.2      *Empirical challenges***

To calculate ancillary benefits and costs over time, one must compare two hypothetical situations. The first is a baseline scenario without any modification of GHG mitigation policy. This is sometimes referred to as “business as usual,” but this term is somewhat misleading since over time, the status quo can change even without modification of GHG policies. The baseline is compared to an even more hypothetical scenario that involves changing the current and future “state of the world” by modifying GHG mitigation. To carry out this exercise in practice means addressing a number of challenges.

How the baseline is defined crucially affects the magnitude of ancillary benefits and costs generated by a change in GHG mitigation policy. The paper by Morgenstern in this proceedings identifies a number of important influences on the baseline. One is the status of non-climate policies. This can be vividly illustrated with two environmental examples. Suppose that even in the absence of climate policy, conventional air pollutants are expected to drop sharply because of trends in policies for the regulation of conventional pollutants. (Note that such a trend requires not just tougher standards over time but also a maintenance or increase in the degree of compliance with those standards.) In this case, we would expect the incremental benefits from a reduction of conventional air pollutants in the wake of tougher GHG controls to be smaller than if the increased GHG controls were being applied to a dirtier baseline environment.

The second example involves the establishment of total emission caps for conventional pollutants, like the cap on sulfur dioxide (SO<sub>2</sub>) from power plants in the U.S. If such a cap is imposed, then a stronger GHG mitigation policy will not have an effect on the total emissions of conventional pollutants unless a much tougher GHG policy is imposed, so tough that it leads to polluters reducing conventional emissions below the legal cap. What would be affected in less stringent cases is the *location* of the conventional emissions, which can have an important effect on the size of exposed populations, etc. This example also illustrates the need for careful cost and benefit accounting when calculating ancillary benefits and costs.

Aside from the interaction of GHG policies and conventional pollutant policies over time, there are several other important elements in specifying the baseline. All the factors driving the evolution of the economic system are included in this list. The state of technology will affect the energy and emissions-intensity of economic activity. The size and location of the population, and the volume and location of total economic output, will affect both the scale of physical impacts on the environment and the risks posed to the population. Finally, the status of natural systems is also part of the baseline; it indicates the sensitivity of humans and ecological resources to changes in conventional pollutants.

Another important set of influences on estimates of ancillary benefits and costs include the scale of analysis, the level of aggregation, and the stringency of the GHG policy being considered. As discussed below, we find that estimates of ancillary health benefits from reduced conventional air pollutants (expressed as dollars per ton of carbon release avoided) tend to get smaller when the analysis shifts from an aggregate perspective to one that considers more carefully the effects of GHG policies on specific sectors at specific locations. These latter analyses appear better able to model the distribution of gains and losses, and the behavioral responses to GHG policies. As for the stringency of GHG policy, we would expect that a stronger GHG program will generate successively smaller increments in ancillary benefits and more ancillary costs as other risks decline relative to baseline levels.

One must remain critical of the assessment of the ancillary impacts themselves. In the area of conventional air pollutants and human health, which has received more research support than others, there nonetheless continues to be considerable uncertainty about how a change in ambient environmental conditions will affect health endpoints (for example, how many fewer cases of disease will result from somewhat cleaner air), and how much society values these changes. We illustrate the effects of these uncertainties below. The uncertainties are especially acute and troubling when one tries to use studies of impacts and valuations from developed countries to assess ancillary benefits in developing countries with lower incomes, different health status and infrastructure, and different cultural norms. Other health and non-health ancillary environmental benefits and costs are even less researched or understood.

Finally, we note that ancillary benefits may not just be physical, but may be economic. One important example is illustrated by returning to the example of the cap on SO<sub>2</sub> from power plants in the U.S. Though there may be no ancillary reductions in emissions of SO<sub>2</sub> as noted, there will be an effect on the cost of compliance under the SO<sub>2</sub> program. Under the cap, a facility that reduces its SO<sub>2</sub> emissions makes emission allowances available for another facility, displacing the need for abatement investment at that facility. If a carbon policy reduces the use of coal in electricity generation, it will lead to a reduction in the demand for SO<sub>2</sub> allowances, thereby avoiding investment in SO<sub>2</sub> abatement. In addition, many studies of the cost of carbon reduction use historically based carbon abatement cost estimates that do not incorporate the effects of the SO<sub>2</sub> cap and thereby *overstate the opportunity cost* of carbon reductions. For instance, the imposition of controls on a conventional pollutant such as SO<sub>2</sub> may reduce the cost advantage that coal has over gas for electricity generation. Layered on top of a control on SO<sub>2</sub>, the reduction of carbon emissions (achieved by substitution from coal to gas) would be less expensive than it would appear were the model to ignore SO<sub>2</sub> controls. Hence, the baseline for comparison of ancillary benefits with costs would be inconsistent, in a potentially important way.

### **3. Adverse human health effects of conventional air pollutants: a review of US studies**

Table 1 summarizes a variety of models and assumptions used to calculate ancillary benefit estimates. References for the estimates are given in Appendix A to this paper. Table 2 summarizes the estimates that are achieved in some of these studies, expressed in the common metric of dollars per ton reduction of carbon emissions. In every case the original studies that produced these data identified a wide range of possible estimates around the midpoint estimate for ancillary benefits per ton of carbon emission reduction that we report. Lower and upper bounds for each estimate vary from the midpoints by a factor of 2 to 10 or more.

Table 1. Description of previous studies of air pollution reduction benefits from greenhouse gas limitations

<i>Study(*) and/or model exercised(**)</i> <i>See Appendix A and Appendix B</i>	<i>Model type</i>	<i>Carbon policy or target</i>	<i>Conventional pollutants and impacts considered</i>	<i>Does baseline include 1990 Clean Air Amendments (including SO<sub>2</sub> cap)?</i>
Goulder (1993)*/ Scheraga and Leary (1993)*	Dynamic general equilibrium	Economy-wide carbon or Btu tax to return total US CO <sub>2</sub> emissions to 1990 levels in 2000 (emissions rise thereafter)	TSP, SO <sub>2</sub> , NO <sub>x</sub> , VOCs, CO, Pb, PM <sub>10</sub> (no secondary particulates or ozone); human health effects only	No (considered in sensitivity analysis)
Jorgenson <i>et al.</i> (1995)*	Dynamic general equilibrium	No specified GHG target; fuel taxes set to internalize conventional air pollution externalities	See entry for Viscusi <i>et al.</i> (1992) below	No
Boyd, Krutilla, Viscusi (1995)*	Static general equilibrium	Energy taxes set either to “optimally internalize” conventional externalities or to exploit all “no regrets” possibilities	See entry for Viscusi <i>et al.</i> (1992) below	No
ICF (1995)*	Partial equilibrium regional model of electricity sector	Voluntary programs under Climate Change Action Plan	CO, TSP, VOCs, NO <sub>x</sub> and PM <sub>10</sub> (SO <sub>2</sub> assumed constant, no secondary particulates); health effects only	Yes
Dowlatabadi <i>et al.</i> (1993)*	Partial equilibrium regional model of electricity sector	Technology policy to improve efficiency and reduce emissions	TSP, NO <sub>x</sub> , and SO <sub>2</sub> (no secondary particulates)	No
Viscusi <i>et al.</i> (1992)*	Valuation only, average for nation	Estimated average damages per unit of emission for various pollutants	TSP, SO <sub>2</sub> , NO <sub>x</sub> , VOCs, CO, Pb, PM <sub>10</sub> (damage from secondary particulates and ozone inferred and attributed to primary pollutants); human health and visibility effects	No
EXMOD (Hagler-Bailly, 1995)**	Detailed electricity sector for NY State, atmospheric transport and valuation	Facility specific emissions and damages; used for sensitivity analysis of other studies	TSP, SO <sub>2</sub> , NO <sub>x</sub> , VOCs, CO, Pb, PM <sub>10</sub> , (secondary particulates and ozone modeled); all human health, visibility and other environmental effects	Yes
PREMIERE (Palmer <i>et al.</i> , 1996)**	Regional electricity sector, atmospheric transport and valuation	Regionally specific emissions and damages; sensitivity analysis of other studies	Only NO <sub>x</sub> (and secondary nitrates) modeled; human health effects only	Yes
<b>HAIKU (Burtraw, <i>et al.</i> 2000)*</b>	Same, detailed electricity sector model	Same	NO <sub>x</sub> and SO <sub>2</sub> (and secondary pollutants modelled); SO <sub>2</sub> cap binding; human health only	Yes, additional reductions
Abt/Pechan (McCubbin <i>et al.</i> 1999)*	Same, for all economic sectors	Same; special attention to avoided abatement costs	SO <sub>2</sub> , NO <sub>x</sub> , PM, CO, O <sub>3</sub> ; Visibility, materials analysed; only health monetized	Yes, additional reductions
Lutter and Shogren (1999)*	Los Angeles	Same, no sensitivity analysis; special attention to avoided abatement costs	Only PM	Yes, additional reductions

**Table 2.** Comparisons of estimates of ancillary benefits per ton of carbon reduction

<b>Source</b> (see Appendix A and B)	<b>Targeted sectors, pollutants, and policy</b> (carbon taxes expressed in 1996 dollars, rounded to the nearest dollar)	<b>Average ancillary benefit per ton carbon reduction</b> (1996 dollars)
(1) HAIKU/ TAF	Nationwide carbon tax of \$25 per ton carbon in electricity sector, analyzed at state level; only health effects from NO <sub>x</sub> changes valued, including secondary particulates, excluding ozone effects. Range of estimates reflect with, and without, NO <sub>x</sub> “SIP call” reductions included in baseline.	\$2-\$5
(2) ICF/ PREMIERE	Nationwide Motor Challenge voluntary program (industry), analyzed at regional level; only health effects from NO <sub>x</sub> changes valued, including secondary particulates, excluding ozone effects.	\$3
(3) Dowlatabadi <i>et al.</i> /PREMIERE	Nationwide seasonal gas burn in place of coal, analyzed at regional level; health effects from NO <sub>x</sub> changes valued using PREMIERE, including secondary nitrates, excluding ozone effects	\$3
(4) EXMOD	Reduced utilization of existing coal steam plant at a suburban New York location; only PM, NO <sub>x</sub> and SO <sub>2</sub> (under emission cap) changes valued (based on 1992 average emissions), including secondary particulates and ozone effects; all health, visibility and environmental effects that could be quantified are included.	\$26
(5) Coal/PREMIERE	Equal percentage reduction in utilization of all existing (1994) coal plants in U.S. analyzed at state level; only health effects from NO <sub>x</sub> changes valued, including secondary particulates and excluding ozone.	\$8
(6) Coal/ PREMIERE/RIA	Equal percentage reduction in utilization of all existing (1994) coal plants in U.S. analyzed at state level; only NO <sub>x</sub> related mortality changes valued, including secondary particulates and excluding ozone, using new EPA RIA estimates of impacts and valuations.	\$26



**Table 2** (continued)

<b>Source</b> (See Appendix A and B)	<b>Targeted sectors, pollutants, and policy</b> (carbon taxes expressed in 1996 dollars, rounded to the nearest dollar)	<b>Average ancillary benefit per ton carbon reduction</b> (1996 dollars)
(7) Abt/Pechan	Carbon taxes of \$30 and \$67 per ton carbon; modeled changes in conventional emissions and concentrations of particulates (no ozone) and changes in health status, visibility and materials damages. Estimates include avoided abatement costs for NO <sub>x</sub> and SO <sub>2</sub> . Attainment areas realize cost savings, nonattainment areas realize air quality improvements. All scenarios include NO <sub>x</sub> "SIP call" reductions in baseline. Estimates reflect outcomes with and without reductions in SO <sub>2</sub> below 1990 Clean Air Act, based on size of carbon tax (high tax leads to net SO <sub>2</sub> reductions).	\$8 - \$68
(8) Goulder/ Scheraga and Leary	Economy-wide carbon tax of \$144 per ton carbon with stabilization at 1990 levels in 2000; human health effects calculated from reduced total emissions of all criteria pollutants, no secondary particulates or ozone.	\$32
(9) Boyd <i>et al.</i>	Economy-wide carbon tax of \$9 per ton carbon; human health and visibility effects calculated from reduced total emissions of all criteria pollutants.	\$39
(10) Viscusi <i>et al.</i>	Equal percentage reduction in utilization of existing (1980 average) coal steam plants nationwide; human health and visibility effects calculated from reduced total emissions of all criteria pollutants.	\$86

Goulder (1993) is one of three modeling efforts that have examined fiscal policies aimed at reducing CO<sub>2</sub> emissions within a general equilibrium model. The model incorporates the intertemporal investment and savings decisions of firms and households, and also accounts for household labor supply decisions. Primary emissions of eight pollutants are modeled (TSP, SO<sub>x</sub>, NO<sub>x</sub>, VOCs, CO, Pb, PM<sub>10</sub> and CO<sub>2</sub>). The model uses fuel-based industry-specific average emission rates, including emissions from mobile sources. Emissions over and above those that can be attributed to fuel use are attributed to output for each industry. Emission factors are held constant at 1990 levels in the initial specification. In sensitivity analysis, SO<sub>2</sub> emissions from the electric utility industry are held constant, in light of the emission allowance trading program, and NO<sub>x</sub>, VOCs and CO emission rates are varied over time to reflect changes in mobile source emissions. NO<sub>x</sub> emission changes from Title IV of the 1990 Clean Air Act are not modeled. There is also no modeling of the economic value of avoided external damages.

The base case in the Goulder model, which ignores the SO<sub>2</sub> cap and other expected changes in emissions, is extended by Scheraga and Leary (1993) to estimate a level of CO<sub>2</sub> emission reductions sufficient to return to 1990-level emissions in the year 2000, about 8.6 percent relative to the base case projection in the model.<sup>130</sup> When a carbon tax is used for this purpose, the emission reductions for conventional pollutants range from 1.4 percent (VOC) to 6.6 percent (NO<sub>x</sub>).

Goulder *et al.* append estimates of the monetary value of avoided health damage culled from a variety of sources, including EPA Regulatory Impact Assessments from the 1980s. They estimate reductions in VOCs, SO<sub>x</sub>, particulates and NO<sub>x</sub> emissions resulting from the carbon tax, yielding benefits in the range of \$300 million to \$3 billion, with benefits about 33 percent greater for a Btu tax. Although the authors do not make this comparison, a rough estimate of the cost of this level of taxation suggests that about one quarter of the cost of the policy is offset by the value of criteria air pollutant reductions.

Jorgenson *et al.* (1995) provides another dynamic general equilibrium model that includes adjustments for projected technical change on an industry basis. Externalities related to global climate change and to criteria air pollutants and acid rain resulting from energy use are modeled. The climate damage values rise over time to reflect the relationship between accumulated greenhouse gases and damages. The 1990 Clean Air Amendments are not reflected in the study. The externality values for reductions in conventional pollutants are unit values adapted from the survey of cost-benefit studies and other research compiled in Viscusi *et al.* (1992), adjusted downward to reduce the estimate of premature mortality associated with sulfur oxides.

These energy related externalities are converted into tax rates under several different scenarios accommodating a range of values for climate and conventional externalities, and they are internalized into prices through *ad valorem* energy taxes, ranging from a 1 percent markup for natural gas to a 197% markup for coal, under their benchmark scenario. The authors also investigate the performance of several strategies for recycling revenue from an energy tax. Their results conform with a “strong form” of the double-dividend hypothesis (Goulder, 1995). This means they find negative (gross) economic costs (that is, positive benefits) from the energy taxes, as measured by equivalent variation defined over goods, services and leisure, when the revenues are used to displace property taxes or capital taxes, even when environmental benefits are not considered.<sup>131</sup> Further, when revenue is recycled by reducing labor taxes, in which case the net economic cost of abatement is positive, the authors find the net benefits of the policy to be positive once reduced conventional pollutant damages are taken into account (not including climate related benefits).

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<sup>130</sup> However, after year 2000 emissions are allowed to increase, which has an implication for the type of abatement measures employed.

<sup>131</sup> This strong finding is contradictory to a large share of recent studies on the subject (Oates, 1995; Goulder, 1995). The main reason for this result is the large economic cost (marginal cost of funds) assumed to result from the use of property or capital taxes to raise government revenues, compared to other studies, as well as the relatively large economic cost of taxes in general represented in the model. However, as noted in the text, they find a less striking result when revenues are recycled to reduce labor taxes, which is the usual assumption.

Boyd, Krutilla and Viscusi (1995) use a simpler general equilibrium model, with land treated as a separate factor of production, to consider *ad valorem* taxes on fuels, with revenues rebated in lump-sum fashion to taxpayers (so there are no gains from recycling revenues to reduce other taxes). Pollutants considered are the same as in Jorgenson *et al.* (1995) and environmental benefit estimates are drawn directly from Viscusi *et al.* (1992). The “optimal” tax levels in the analysis are defined as those that maximize the sum of benefits from reducing conventional environmental externalities (excluding any benefits from reducing carbon emissions) less the economic costs of the tax.

In the base case the optimal carbon emission reductions are 0.19 billion tons (about 12 percent of total emissions). The authors report the optimal *ad valorem* tax on coal is about 45 percent, comparable to a \$8/ton carbon charge.<sup>132</sup> The authors also identify the “no regrets” level of reduction in the analysis as the point at which net benefits from internalizing conventional environmental externalities drop to zero. This is equal to 0.5 billion tons (a 29 percent reduction), which would be achieved with a \$13 tax per ton carbon (leading to a 54 percent *ad valorem* tax on coal). In the case of a higher substitution elasticity between energy and other factors of production, the no regrets level of carbon reduction is estimated to be about 0.8 billion tons (49 percent reduction).

Two other modeling efforts are based on frameworks that include considerable detail about the electricity industry. Holmes *et al.* (1995) (ICF) use the DEGREES model to examine four out of approximately 50 actions identified in the Climate Change Action Plan announced by the Clinton Administration in 1993, and the impact these actions would have on electricity demand, generation, and associated emissions. These actions include expansion of the Green Lights Program, energy efficient electrical motor systems (Motor Challenge), improvement of hydroelectric generation, and reform of electricity transmission pricing. Pollutants modeled include NO<sub>x</sub>, SO<sub>2</sub>, CO, TSP, VOCs, and PM<sub>10</sub>.

The study examines the change in emissions on a geographic basis, according to North American Electric Reliability Council (NERC) Regions. Regional variation in emission changes stems in large part from the variation in technologies providing electricity at the margin and that would be affected by each of the actions. In some regions of the country, for example, gas facilities would be more likely to be displaced while in other regions coal facilities may be displaced, and these fuels and technologies typically have very different emission rates. The study is unique because it examines changes on a seasonal and time-of-day temporal basis, by modeling changes in the electricity load duration curve and facility operation. In addition, the study is the most comprehensive in the consideration of changes in emission rates already destined to occur due to provisions in Title IV of the 1990 Clean Air Act Amendments. The study suggests that SO<sub>2</sub> emissions will be approximately invariant to the actions that are studied, though the timing of emission reductions under Title IV may be affected by the policies that were evaluated. Baseline NO<sub>x</sub> emissions are also projected to fall due to the requirements of Title IV.

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<sup>132</sup>

We have difficulty replicating their calculations regarding the carbon charges.

Dowlatabadi *et al.* (1993) employ another detailed model of the electric utility system called the Energy Policy Assessment model to assess emission changes at the regional level. This modeling effort was based on a 1987 plant inventory, and it did not include changes resulting from the 1990 Clean Air Act Amendments. Pollutants that were modeled in addition to CO<sub>2</sub> were SO<sub>2</sub>, NO<sub>x</sub> and TSP. In common with the ICF model, this model reported results by NERC region. The model was used to consider technology including seasonal gas burning; use of externality adders in dispatch of facilities; extension of the life of nuclear facilities; elimination of federal subsidies; and improvement of the efficiency of electricity distribution transformers.

A main contribution of the study was to illuminate the potential importance of double-counting of emission changes when individual policies affect the same endpoints. The emission changes from these policies are not additive because the policies taken separately would each capture the same low-cost substitution opportunities that would not be available in similar degree to the policies taken as a group. The ratio of the emission changes for NO<sub>x</sub> for the strategies considered collectively is 11 percent less than the sum of emission changes when the policies are considered separately in the short run scenario. The study also illuminates potential perverse effects from technology policy. For example, the NO<sub>x</sub> emissions that could result as people switch to gas use for home and water heating in response to changes in electricity prices could be greater than the NO<sub>x</sub> emissions from centralized electricity generation sources providing the same energy services. In addition, emissions from gas use in the home are distributed throughout a metropolitan area. This could have greater environmental damages than emissions from sources more distant from population centers, potentially offsetting some of the ancillary benefits from carbon policies.

McCubbin *et al.* (1999) (Abt/Pechan) is a detailed analysis similar to the HAIKU/TAF analysis that we characterize below as “best practice.” McCubbin *et al.* assume the implementation of carbon taxes and estimate changes in energy consumption by region and sector of the economy. These changes are translated into changes in emissions, and then translated into changes in concentrations of particulates using a source-receptor matrix. Finally, these are mapped into changes in health status and valued in monetary terms. The modeling steps are those used in other detailed, peer-reviewed studies conducted by the US EPA. McCubbin *et al.* paid careful attention to revisions in the US air quality standards in constructing their baselines. The study accounted for reductions in compliance costs for achieving ambient air quality standards in regions of the country that are in attainment of air quality standards, as well as improvements in air quality and health status in regions that are in nonattainment.

A limitation of the study is that the total carbon reductions that are achieved under the tax is not reported, probably because of the political sensitivity of the question of the cost of achieving carbon reductions in the US. This omission makes estimation of ancillary benefits per ton of carbon reduced difficult. In some cases additional reductions of SO<sub>2</sub> in the baseline are achieved through new air quality standards that reduce the annual cap. However, when they are not achieved in the baseline, then the carbon policy yields reductions because annual emissions fall below the cap established in the 1990 Clean Air Act. This yields the high end estimates they present. We note that the proportion of SO<sub>2</sub> reductions that are achieved relative to carbon reductions is greater than that in HAIKU/TAF.

One other study that we do not include in our review of estimates is Lutter and Shogren (1999). This study offers an analytical description of how changes in carbon emissions affect the emissions of other pollutants and includes an accounting of the reduction in compliance costs in achieving air quality standards for conventional pollutants. This point is embodied in some of the other papers we review. Lutter and Shogren illustrate this point in the special context of Los Angeles where the compliance cost of strict new ambient standards would be especially dramatic. In this case, carbon reductions yield large ancillary benefits of around \$300 per ton from reduced compliance costs. The authors point out that ancillary benefits would have an effect on whether a nation should choose to reduce emissions domestically or seek to acquire permits in a trading program.

#### **4. The HAIKU/TAF model: An illustration of best practice**

The study by Burtraw *et al.* (2000), which uses the HAIKU/TAF integrated framework, is one we believe illustrates best practice (as noted, the methods are similar to those in McCubbin *et al.* (1999)). Burtraw *et al.* exercise an electricity market model to predict changes in nitrogen oxides (NO<sub>x</sub>), sulfur dioxide (SO<sub>2</sub>), and mercury from moderate carbon policies. These changes are fed through an atmospheric transport and health model to predict changes in health status, and to characterize these changes in monetary terms. Additional savings accrue from reduced investment in SO<sub>2</sub> abatement in order to comply with the SO<sub>2</sub> emission cap due to the shift away from coal for electricity generation under a carbon tax.

The study directly addresses three methodological issues that are important to the consideration of how GHG mitigation could yield ancillary benefits. These include the characterization of changes in emissions, the characterization of health benefits, and the baseline against which these changes are measured.

##### **4.1 Emissions**

Burtraw *et al.* focus exclusively on air emissions and their potential health effects, which as noted account for 85 percent or more of the quantifiable environmental concerns in previous studies of the electricity sector. Also, they focus exclusively the electricity sector. While the electricity sector is responsible for one-third of carbon emissions presently, the EIA projects that this sector will be responsible for as much as three-quarters of CO<sub>2</sub> emission reductions in the U.S. under potential implementation of the Kyoto Protocol (EIA, 1998). Hence, this sector will be especially important as the least expensive and likely source of reductions under moderate reduction scenarios.

##### **4.2 Health effects**

This paper uses the “damage function approach” to focus on estimating the social cost of electricity generation from facilities examined on an individual basis. The approach has been used in recent analyses of environmental impacts of electric power plant siting and operation in specific geographic locations (Lee *et al.*, 1995; EC, 1995; Rowe *et al.*, 1995). The approach involves an atmospheric transport model linking changes in emissions at a specific geographic location with changes in exposure at other locations. Concentration-response functions are used to predict changes in mortality and a number of morbidity endpoints, and these changes are valued in monetary terms drawing on estimates from the economics literature.

The model accounts for expected changes in population, and for expected changes in income that affect estimates of willingness to pay for improvements in health status. These are important considerations since population and income trends have greatly outstripped energy prices over the last century. U.S. population is expected to grow by 45 percent over just the next fifty years, which coupled with expected income growth, suggests that there will be greater exposure to a given level of pollution and consequently greater benefits from reducing that pollution (Krutilla, 1967). This demographic consideration suggests that the reported values for conventional pollutants in previous studies underestimate damage in future years, if all other things are equal.

### 4.3 *The baseline*

In a static analysis the baseline can be treated as the status quo, but since climate policy inherently is a longer-term effort, questions arise about projecting energy use, technology investments, and emissions of GHGs and criteria pollutants with and without the GHG policy. The issue is confounded because of ongoing changes in the standards for criteria air pollutants. If one proceeds on the basis of historical standards and ignores expected changes in the standards, the ancillary benefit estimate will overstate environmental savings. Indeed, historical emission rates may be ten times the rates that apply for new facilities. In addition, the recent tightening of standards for ozone and particulates and associated improvements in environmental performance over time imply that benefits from reductions in criteria air pollutants resulting from climate policies will be smaller in the future than in the present.

Burtraw *et al.* include alternative baselines for NO<sub>x</sub> controls beyond Phase II of Title IV of the 1990 Clean Air Act Amendments. One baseline adds a cap and trade program for the summer months for the Ozone Transport Commission member states in the northeast. This trading program began in 1999. An alternative adds a cap and trade program for a larger region that includes all the eastern states in the Ozone Transport Rulemaking Region (the so-called “SIP call region”) affected by the EPA’s September 1998 proposed rule regarding NO<sub>x</sub> emissions.

Another important example of a regulatory baseline is the cap on SO<sub>2</sub> emissions from electricity generation in the U.S. A consequence of the current emissions cap is that aggregate SO<sub>2</sub> emissions from electric utilities (the major source category in the U.S.) are not likely to change much as a result of moderate GHG emission reductions such as we describe in this paper. Only if climate policies are sufficiently stringent that utilities substitute away from coal in significant fashion and the long-run annual level of SO<sub>2</sub> emissions is less than the annual emissions cap would ancillary benefits from further reductions in SO<sub>2</sub> be achieved.

Many previous studies use historical SO<sub>2</sub> emission rates and do not incorporate the SO<sub>2</sub> emission cap, and hence they overstate the ancillary benefits that may be achieved, at least by moderate climate policies. By the same token, however, historically based CO<sub>2</sub> abatement cost estimates that do not incorporate the effects of the SO<sub>2</sub> cap overstate the opportunity cost of CO<sub>2</sub> reductions. For instance, the imposition of controls on a conventional pollutant such as SO<sub>2</sub> may reduce the cost advantage that coal has over gas for electricity generation. Layered on top of a control on SO<sub>2</sub>, the reduction of CO<sub>2</sub> emissions (achieved by substitution from coal to gas) would be less expensive than it would appear were the model to ignore the SO<sub>2</sub> controls.

Further, there is an ancillary economic saving associated with CO<sub>2</sub> reductions, even with a binding SO<sub>2</sub> emissions cap. Under the cap, a facility that reduces its SO<sub>2</sub> emissions makes emission allowances available for another facility, displacing the need for abatement investment at that facility.

Burtraw *et al.* assume the SO<sub>2</sub> cap is binding and hence they do not anticipate ancillary benefits from changes in SO<sub>2</sub> emissions for moderate policies. However, they do anticipate reduced costs of compliance with the SO<sub>2</sub> cap to result as a consequence of climate policies, and these savings are reflected in expected change in electricity prices and consumer and producer surplus. If new standards regarding NO<sub>x</sub> emissions from power plants take the form of a cap and trade program analogous to the SO<sub>2</sub> program, but applied only during the summer months, then further emissions in NO<sub>x</sub> will be less than under a performance standard. However, in this case we would expect a greater ancillary economic saving due the avoided abatement investment for NO<sub>x</sub> controls, analogous to the avoided abatement investment for SO<sub>2</sub> controls under the SO<sub>2</sub> cap.

Finally, the issue of baselines is complicated further by the changes in the regulation of the electricity industry. At the time of this writing (summer 2000) over half of the US population reside in states that have committed themselves to a path of restructuring that would culminate in a move away from cost of service pricing for electricity and toward market-based, marginal cost pricing. This change has the potential and is expected by many to affect dramatically the emissions of various pollutants. Burtraw *et al.* adopt a cautious assumption regarding the future regulation of the industry by assuming that traditional average cost pricing continues in effect in regions that have not committed to marginal cost pricing by 2000.

#### **4.4      *The models***

The study employs an electricity market equilibrium model called HAIKU to simulate market equilibrium in regional electricity markets and inter-regional electricity trade, with a fully integrated algorithm for NO<sub>x</sub> and SO<sub>2</sub> emission control technology choice. The model simulates electricity demand, electricity prices, the composition of electricity supply, inter-regional electricity trading activity among NERC regions, and emissions of key pollutants such as NO<sub>x</sub>, SO<sub>2</sub>, mercury and CO<sub>2</sub> from electricity generation. Investment in new generation capacity and retirement of existing facilities are determined endogenously in the model, based on capacity-related “going forward costs.” Generator dispatch in the model is based on minimization of short run variable costs of generation.

Two components of the HAIKU model are the Intra-regional Electricity Market Component and the Inter-regional Power Trading Component. The Intra-regional Electricity Market Component solves for a market equilibrium identified by the intersection of electricity demand for three customer classes (residential, industrial and commercial) and supply curves for each of three time periods (peak, middle and off-peak hours) in each of three seasons (summer, winter, and spring/fall) within each of 13 NERC regions. The Inter-regional Power Trading Component solves for the level of inter-regional power trading necessary to equilibrate regional electricity prices (gross of transmission costs and power losses). These inter-regional transactions are constrained by the assumed level of available inter-regional transmission capability as reported by NERC.

Technical parameters are set to reflect midpoint assumptions by the EIA and other organizations regarding technological change, growth in transmission capacity, and a number of other factors. Most new investment is in conventional technologies including integrated combined cycle natural gas units and gas turbines. Fuel supply is price responsive according to a schedule derived from EIA models.

To estimate the potential for carbon emission reductions, Burtraw *et al.* impose a tax on all emissions in the industry. This tax is collected through the price of electricity and affects dispatch and investment decisions. Tax levels of \$10-\$50 per metric ton of carbon are far below the EIA's estimated tax of \$348 per metric ton carbon required to achieve Kyoto budgets in 2010 in the absence of international trading.

The changes in emissions of NO<sub>x</sub> are fed into the Tracking and Analysis Framework (TAF). TAF is a nonproprietary and peer-reviewed model constructed with the *Analytica* modeling software (Bloyd *et al.*, 1996).<sup>133</sup> TAF integrates pollutant transport and deposition (including formation of secondary particulates but excluding ozone), visibility effects, effects on recreational lake fishing through changes in soil and aquatic chemistry, human health effects, and valuation of benefits. All effects are evaluated at the state level and changes outside the U.S. are not evaluated.

Health effects are characterized as changes in health status predicted to result from changes in air pollution concentrations. Impacts are expressed as the number of days of acute morbidity effects of various types, the number of chronic disease cases, and the number of statistical lives lost to premature death. The health module is based on concentration-response (C-R) functions found in the peer-reviewed literature. The C-R functions are taken, for the most part, from articles reviewed in the U.S. Environmental Protection Agency (EPA) Criteria Documents (see, for example, USEPA 1995, USEPA 1996b). The Health Effects Module contains C-R functions for PM<sub>10</sub>, total suspended particulates (TSP), sulfur dioxide (SO<sub>2</sub>), sulfates (SO<sub>4</sub>), nitrogen dioxide (NO<sub>2</sub>), and nitrates (NO<sub>3</sub>). The change in the annual number of impacts of each health endpoint is the output that is valued. In this exercise inputs consist of changes in ambient concentrations of NO<sub>x</sub>, and demographic information on the population of interest. The numbers used to value these effects are similar to those used in recent Regulatory Impact Analysis by the USEPA.

#### 4.5 Results

We use the HAIKU/TAF model to evaluate two scenarios, and results from the analysis are presented in Table 3. (See Burtraw *et al.* 2000 for a fuller discussion.) In the first scenario, identified as Baseline OTC, NO<sub>x</sub> standards implemented in the 1990 Clean Air Act are maintained, except for an additional cap and trade program that applies during the five month summer season in the Northeast states. A carbon tax of \$23 per metric ton of carbon in the year 2010 would yield ancillary benefits from reductions in NO<sub>x</sub> of \$7 for each ton of carbon reduced (1996 dollars). The primary category of these benefits is mortality. Morbidity benefits are also significant. Previous analysis (Burtraw *et al.*, 1998) using TAF indicates that the value of visibility improvements are about the same order of magnitude as morbidity benefits, but these results are not included here.

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<sup>133</sup> Each module of TAF was constructed and refined by a group of experts in that field, and draws primarily on peer reviewed literature to construct the integrated model. TAF is the work of a team of over 30 modelers and scientists from institutions around the country. As the framework integrating these literatures, TAF itself was subject to an extensive peer review in December 1995, which concluded that "TAF represent(s) a major advancement in our ability to perform integrated assessments" and that the model was ready for use by NAPAP (ORNL, 1995). The entire model is available at [www.lumina.com/taflist](http://www.lumina.com/taflist).



Measured against the Baseline OTC case, an \$83 tax increases ancillary benefits in the aggregate but has little effect on the value per ton of carbon, which increases slightly to \$8. The quantity of carbon emission reductions that are achieved by an \$83 tax is proportionately less than that achieved by a \$23 tax, which illustrates that the marginal abatement cost curve for carbon reductions is convex over this range. The proportional change in NO<sub>x</sub> emissions is also less than the change in the tax rates, but it is not strictly tied to changes in carbon emissions. The difference in the ratios of NO<sub>x</sub> and carbon reductions stem from many factors including cost thresholds for new investment and retirement, and from the geographic location of changes in emissions. In other scenarios the benefits per ton fall slightly or stay relatively constant with different levels of a carbon tax.

Table 3. **Ancillary health benefits from reductions in NO<sub>x</sub> emissions resulting for various carbon taxes in the electricity sector in 2010 using HAIKU/TAF (1996 dollars)**

<b>Level of Carbon Tax</b> (\$/metric ton)	<b>Baseline - OTC</b>		<b>Baseline - SIP</b>	
	<b>23</b>	<b>83</b>	<b>23</b>	<b>83</b>
<b>Emission Reductions</b> (metric tons)				
Carbon (millions)	79	214	88	215
NO <sub>x</sub> (thousands)	874	2586	974	2564
<b>NO<sub>x</sub> Related Health Benefits</b> (million dollars)				
Morbidity	115	368	157	388
Mortality	437	1,382	585	1460
Total	552	1,750	742	1848
<b>NO<sub>x</sub> Related Health Benefits per Ton Carbon</b> (dollars)				
Morbidity	1.5	1.7	1.8	1.8
Mortality	5.5	6.5	6.6	6.8
<b>Total</b>	<b>7.0</b>	<b>8.2</b>	<b>8.4</b>	<b>8.6</b>

We can examine how much of the increase in NO<sub>x</sub> benefits is related to locational differences in generation by comparing the benefits per ton of NO<sub>x</sub> reduction under the two levels of tax. From Table 3, the benefits from a reduction in NO<sub>x</sub> emissions fall from \$676 per ton under an \$83 carbon tax, to \$632 per ton of NO<sub>x</sub> at a \$23 carbon tax. This reduction in the benefit per ton of NO<sub>x</sub> reduction is due in part to difference in the location of reductions and generation technology. In essence, this means that the additional sources reacting to the higher carbon tax have different emission rates for NO<sub>x</sub> and are located in areas where the conversion of NO<sub>x</sub> to nitrates is less efficient, or where fewer people are being exposed to the nitrate concentrations, or both. Taken together, the nonlinearity in emission reductions and in the benefits of those reductions provides an indication of the importance of using a regionally disaggregated model to investigate this issue, unlike some of the previous studies that are discussed below.

The emission reductions are achieved by a dramatic shift from generation with coal to generation with gas. Table 4 indicates that generation by coal under the \$83 tax falls to 39 percent of that in the baseline. In contrast, generation by natural gas increases to 158 percent of that in the baseline. These changes are achieved in 2010 and result from a policy that is implemented in 2001. The results would differ if the policy were implemented later.

Table 4. **The ratio of generation under tax to generation under the baseline for each respective scenario**

<b>Level of Carbon Tax</b> (\$/metric ton)	<b>Baseline - OTC</b>		<b>Baseline - SIP</b>	
	<b>23</b>	<b>83</b>	<b>23</b>	<b>83</b>
<b>Gas Generation</b>	1.18	1.58	1.22	1.60
<b>Coal Generation</b>	.79	.39	.77	.40

In an alternative scenario, we consider implementation of a summertime NO<sub>x</sub> cap and trade program in the larger eastern US (the SIP call region) during the summer months. These results are labeled the Baseline - SIP scenario in Table 3. The estimate of ancillary benefits per ton of carbon reduction under an \$83 tax is \$8.6 in this setting, and under a \$23 tax it is \$8.4. Compared to the results for the previous case, the Baseline - SIP scenario yields a less of a reduction in NO<sub>x</sub> under an \$83 tax but greater NO<sub>x</sub> related health benefits and greater benefits per ton of carbon reduction. Again, this illustrates the importance of specificity in modeling the location of carbon reductions and the technology choice. Table 4 reports differences in generation for this scenario as well.

SO<sub>2</sub> emissions are presumed to be unchanged in these scenarios for the size of carbon taxes we consider. However, ancillary savings associated with reduced investment in SO<sub>2</sub> abatement that results from decreased use of coal in electricity generation would add roughly an additional \$3 per ton of carbon emission reductions.

As noted, the results pertain to the year 2010 from policies implemented beginning in 2001. Hence, the cost of the policy is incurred earlier than 2010, but also there are reductions achieved before 2010. The schedule of costs and benefits over time is important in calculating a present discounted value of ancillary benefits, and is discussed in Burtraw *et al.* Also, the estimates developed under these scenarios correspond to carbon taxes that reflect the expected marginal cost of carbon abatement. The *average* cost per ton of carbon reduced will be less than the tax on a per ton basis, and hence ancillary benefits may come close to justifying the carbon tax at moderate levels. Burtraw *et al.* discuss the cost of the policies in terms of changes in consumer and producer surplus in a manner that can be directly compared with ancillary benefits, as well as direct benefits of GHG mitigation, though the latter is notoriously controversial and difficult to quantify.

## 5. Interpretation of the estimates

### 5.1 General observations

In this section we attempt to compare previous ancillary benefit estimates along a common metric, by expressing mid-value estimates per ton reduction in carbon emissions where such a calculation can be made given the information from the studies. In preparing Table 2, we have supplemented the per-ton of carbon estimates directly available from the studies in Table 1 in two ways. The results of the Holmes *et al.* (1995) study could be used for a geographic analysis of atmospheric transport of pollution and exposure of the population, and economic valuation of emission changes. However, this was not attempted in the report. To supplement this analysis, we fed the emission changes into PREMIERE, a model that employs a reduced-form atmospheric transport model linked to monetary valuation of health impacts at a NERC region level.<sup>134</sup> Similarly, we supplement the Dowlatabadi *et al.* analysis listed in Table 1 by feeding predicted emission changes into PREMIERE. These calculations are described in more detail in Appendix B.

The comparison of estimates is reported in Table 2, which indicates a large variation across studies in their mid-range ancillary benefit estimates. Note that in every case there is a wide range of values around the mid-range estimate, which we do not report. Lower and upper bounds for each estimate range varies from its midpoint by a factor of 2 to 10 or more. Several differences in the models account for the bulk of the differences in the results. One is the modeling of criteria pollutant emissions reductions. The general equilibrium models have the advantage in predicting emissions changes in the future because they can account for changes in the quantity of electricity demand and substitution among technologies. However, they are likely to have less accuracy for near-term emission changes because they have less detailed modeling of technology.

However, longer-term future changes in pollution standards are not accounted for in any of the studies assessing GHG policies that we discuss below (including our own). As a practical matter, this means our estimates of ancillary benefits should be considered more reliable for near-term GHG policies than for policies that are actually implemented in the 2008-2012 “commitment period” identified in the Kyoto Protocol. Other things equal (which in practice is not the case), we would expect progress toward improved air quality in the U.S. to reduce ancillary benefits below the amounts shown in Table 2.

The estimation and valuation of effects from emission changes varies among the studies. It is relatively weak in the general equilibrium models, which do not treat locational differences. More aggregated analyses calculate total emissions changes and apply a single unit value to value the avoided health impacts. In contrast, disaggregated models can more precisely model the location of emissions, their transport through the atmosphere, and the exposure of affected populations. These analyses show that benefits do not have a simple proportional relationship to reduced emissions. Sensitivity analyses show that the above-mentioned aspects are important influences on ancillary benefits, so the greater precision with which they are calculated in disaggregated models give us greater confidence in these results.

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<sup>134</sup> Palmer et al. (1996). PREMIERE is a derivative of the Tracking and Analysis Framework (TAF) discussed above.

Another reason for the difference in per ton benefit estimates is differences in sectoral coverage and coverage of pollutants or impacts. For example, the estimates presented range from a small voluntary program affecting the electricity sector to estimates for the economy as a whole.

The treatment of the SO<sub>2</sub> cap represents another important distinction among the studies. When the cap is binding, emission reductions in one location are made up in another, but emissions at one location are likely to reduce the need for investment in SO<sub>2</sub> abatement at another location. This is usually not considered in cost estimates for CO<sub>2</sub> reduction. For example, our estimates using PREMIERE and EXMOD (discussed below) incorporate a secondary benefit of about \$3 per ton of carbon reduction from avoided investment in SO<sub>2</sub> abatement stemming from reduced utilization of coal. This benefit is likely to be considerably smaller than the health benefit that would be induced if total SO<sub>2</sub> emissions were reduced by a GHG policy, leading to a reduction in fine sulfate particles implicated in increased premature mortality (Burtraw *et al.*, 1998).

An important corollary of this observation is that the marginal ancillary benefits from a small reduction in GHGs are likely to differ from the marginal benefit from the last unit of GHG reduction in a more aggressive program of aggregate GHG control. Even if the underlying atmospheric transport and health effects models are essentially linear, as the studies presented here implicitly or explicitly assume, there will be a threshold at the point where GHG control has made the SO<sub>2</sub> cap no longer binding. Beyond this point, health benefits from additional net reductions in SO<sub>2</sub> will accrue.

## 5.2 *Comments on the studies*

With these thoughts in mind, if one wants to identify the ancillary benefit per ton of carbon reductions for a *modest* carbon abatement program *given the presumed baseline conditions*, we would place more confidence in the first five estimates in Table 2. All of these estimates reflect the impact of GHG reductions in the electricity sector. These estimates reflect the most detailed methodologies, including locational differences in emissions and exposures, and they take into account the role of the SO<sub>2</sub> cap in limiting ancillary benefits. Note that these estimates suggest modest ancillary benefits (less than \$8 per ton carbon) for studies averaging over the United States as a whole from electricity sector GHG reductions, though benefits could be significantly higher in certain areas (New York).

The first three studies in Table 2 indicate that subtle aspects of behavioral responses to policies tend to mitigate the desired emission reductions.<sup>135</sup> The HAIKU/TAF example demonstrates that the location of carbon emissions and the choice of technology for generation in response to a carbon tax will vary over time and space. This leads to variation in the value of ancillary health benefits per ton of carbon reduced. Nonetheless, the values in the two scenarios we compare are within a fairly small range around \$8 per ton of carbon reduction.

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The Dowlatabadi *et al.* estimates may exaggerate this effect because they reflect the capital stock circa 1987 and do not reflect improvements in gas technologies.

The ICF/PREMIERE example also estimates health benefits from changes in NO<sub>x</sub> emissions and transport (excluding ozone effects) for a voluntary policy. This estimate is low due to the fact that some of the reduced electricity generation resulting from energy efficiency improvements will come from natural gas units that have lower emission rates for NO<sub>x</sub> than do coal units and hence fewer ancillary benefits obtain. Dowlatabadi *et al.*/PREMIERE reflects a seasonal (summer) burn of natural gas in place of coal, and models health benefits from changes in NO<sub>x</sub> emissions and their transport (excluding ozone effects). These results are low because increased emissions of NO<sub>x</sub> from gas offsets somewhat the reductions from coal.<sup>136</sup>

The EXMOD estimate is greater than the three preceding ones because it does not account for the bounceback effect that may result from increased utilization of another technology such as natural gas to replace coal utilization, and because it is set in a densely populated area. The EXMOD estimate uses average emission rates from an existing coal steam plant in a relatively densely populated suburban area in New York State, with a reduced-form model of atmospheric dispersion, exposure and valuation, and it accounts for SO<sub>2</sub> trading as discussed above. This estimate includes health damages from airborne exposure to particulates, NO<sub>x</sub> (including ozone) and changes in the location of SO<sub>2</sub> emissions under the cap, holding total emissions constant. Collectively these are calculated to be 90-96 percent of the damage from conventional pollutants through all environmental pathways.

The fifth estimate, Coal/PREMIERE, is comparable to the fourth, except that it is applied on a weighted average national basis. This example considers a 1 percent reduction in utilization of coal fired electricity generation and calculates changes in CO<sub>2</sub>, SO<sub>2</sub> and NO<sub>x</sub> emissions at the regional level for use in PREMIERE. The benefits per ton carbon reflect only changes in NO<sub>x</sub>, excluding both ozone impacts and SO<sub>2</sub> changes (due to the cap). About 65 percent of the NO<sub>x</sub> related benefits result from decreased mortality.<sup>137</sup>

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<sup>136</sup> We ignore the Dowlatabadi et al. estimates for SO<sub>2</sub> because they do not model the allowance trading program, and we ignore the reduction in TSP because it is negligible.

<sup>137</sup> SO<sub>x</sub> changes are not included due to the SO<sub>2</sub> cap, but they would amount to \$87 per ton carbon were emissions not made up through the trading program.

The sensitivity of conclusions to the valuation of damages is illustrated by comparing the PREMIERE and EXMOD estimates to the sixth estimate in Table 2, which uses assumptions drawn from a recent Draft Regulatory Impact Analysis (RIA) for new particulate and ozone standards (USEPA, 1996). The Coal/PREMIERE/RIA example considers the same change in emissions, with atmospheric transport calculated with PREMIERE, but with an assumption that the mortality coefficient used in the RIA for PM<sub>2.5</sub> applies to nitrates. The RIA also places greater weight on one study, Pope *et al.* (1995), leading to greater estimates of long-term mortality than does PREMIERE, which treats this as a high estimate in a distribution of possible estimates. Finally, the valuation of mortality effects in the RIA is about 1.5 times that in PREMIERE (USEPA, 1996). On net this approach yields a valuation of mortality impacts from NO<sub>x</sub> changes (excluding ozone impacts) of three times that from PREMIERE.<sup>138</sup> However, given the controversy surrounding these specific assumptions and our belief that these assumptions overstate ancillary benefits, we put less stock in it.<sup>139</sup>

The seventh study, Abt/Pechan (McCubben *et al.* 1999), is another detailed analysis and it achieves estimates very similar to other studies when exercised under similar scenarios. This is reflected in the low end of the estimates cited in Table 2. The low end of the estimates does not include benefits from reductions in SO<sub>2</sub> under the assumption that a cap and trade program is operative. However, the cap is one-half of that mandated in the 1990 Clean Air Act Amendments. In the scenario yielding the high end estimate, the SO<sub>2</sub> cap is left at the levels in the 1990 legislation in the baseline, and in this case the study achieves high benefit estimates because the carbon tax that is modeled makes this constraint slack, resulting in reductions in SO<sub>2</sub> below the cap. The results differ from those in HAIKU/TAF, which does not find reductions in SO<sub>2</sub> under a comparable (larger) carbon tax because the cap remains binding at the levels in the 1990 legislation.

The next three estimates are the results from general equilibrium modeling. We feel the base on which valuations in the general equilibrium models have been constructed is narrow, as illustrated by the fact that the estimates in Boyd *et al.*, like those in Jorgenson *et al.*, are based on Viscusi *et al.* (The Jorgenson *et al.* 1995 estimate is expressed as a percentage of carbon tax revenue, and GHG reductions are not reported, so it is not shown in Table 2.) Viscusi *et al.* report values that reflect a reduction in secondary pollutants absent geographic resolution, and the authors report the value per ton of secondary pollutant. We convert this using their source data to dollars per kilowatt-hour of generation from a generic existing coal plant in the late 1980s, and then convert to dollars per ton carbon reduction reflecting an assumption that the relative emission rates remain constant. The Goulder/Scheraga-Leary valuation is based on a different review of EPA Regulatory Impact Assessments from the 1980s, which provides a little more breadth to the analyses as a group.

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<sup>138</sup> One can also ask how the use of a reduced form version of the Advanced Statistical Trajectory Regional Air Pollution (ASTRAP) for modeling atmospheric transport in PREMIERE compares with the use of Regional Acid Deposition Model (RADM), which is the model used in the Draft RIA. Burtraw *et al.*, 1997 compared the two directly and find RADM yields valuation numbers about 50 percent less than ASTRAP when considering sulfates, but no comparison of nitrates was made.

<sup>139</sup> One recent analysis (Krupnick *et al.* 2000) suggests that the value of reducing premature mortality, when considered in the context of reduction in conventional air pollutants, is significantly lower than the usual estimates applied in all of the studies reported here. On the other hand, there is some evidence of a stronger link between ozone concentrations and premature mortality than is represented in the existing studies considered here.

We have not addressed previous European studies, many of which are described elsewhere in this volume, but some comparison of the estimates is useful. Ekins (1996) provided a review of the first generation of European study and arrived at a point estimate of about \$272 (1996 dollars) per ton in total benefits, based on his analysis and evaluation of the half dozen or so studies he reviews. About half of the estimated benefits would come from reduced sulfur emissions, and this estimate does not take into account the SO<sub>2</sub> emission reductions that will result from the signing of the European Second Sulphur Protocol in 1994. Following the reasoning provided by Ekins and the studies he reviews, we reduce this estimate to account for the Second Sulphur Protocol, to arrive at a range of \$40-\$85 per ton (1996 dollars) for sulfur benefits only.<sup>140</sup> Adding in benefits of about \$126 per ton from reduced emissions of other pollutants increases this to a range of \$166-\$211, with a mid-value of \$188. This value is relatively high, which may reflect the aggregate level of modeling in these studies, different assumptions about health epidemiology, greater population density in Europe, and the ecological effects resulting from on-shore atmospheric transport of sulfur, in contrast to off-shore transport in the eastern U.S.<sup>141</sup>

## 6. Conclusions

### 6.1 *The scale of ancillary benefits*

How does one make sense of the welter of estimates in Table 2? The first point is that firm conclusions are all but impossible to draw at present, given the current state of knowledge. Accordingly, we do not believe it is possible at this time to identify a single numerical “best estimate” of benefits per ton carbon reduced for any particular GHG limitation, let alone for all possible GHG limitations. As discussed in more detail below, we believe there are modest but nonetheless important ancillary benefits per ton of carbon emission reduction that would result from a modest level of GHG control, and that the benefits may be more than modest in certain locations (those with denser populations and greater exposures to damaging criteria pollutants). The benefits per ton of carbon reduction could be larger with a greater degree of GHG control, though it is difficult to gauge by how much.

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<sup>140</sup> Ekins adjusts his point estimate to account for planned reductions in sulfur emissions stemming from the Second Sulfur Protocol signed in 1994 but not yet implemented, to arrive at an estimate of \$25 for SO<sub>2</sub> related benefits per short ton in the UK only if realized as additional emission reductions, or \$42 if realized as avoided investments in abatement. Note that the latter figure is far larger than the \$3/ton for the U.S. that we estimate. Ekins also notes benefits in the UK from reduced SO<sub>2</sub> emissions range from 35-81 percent total (European) secondary benefits applicable to changes in emissions from the UK. We infer the range of \$33-\$71 (in 1990 dollars) if benefits are realized through additional emission reductions.

<sup>141</sup> See Krupnick and Burtraw (1997) for a related discussion.

In identifying the large uncertainties surrounding current estimates of ancillary benefits, we have focused especially on the location of emissions reductions, the role of the SO<sub>2</sub> emissions cap, and the means by which emissions reductions are achieved (e.g., voluntary versus involuntary measures, and comprehensive measures versus measures that allow increases in emissions from uncovered sources). Additional factors include basic questions about the baseline against which to measure the effects of policy options (e.g. trends in criteria pollutant emissions), atmospheric modeling of the transport of these emissions, the incidence of adverse effects of these emissions, and the economic valuation of avoided adverse impacts. The literature provides little in the way of estimates for ancillary benefits other than those associated with the electricity sector.<sup>142</sup> A more reliable and comprehensive set of estimates must await the analysis of how GHG abatement policies would affect other emissions sources, among other advances in knowledge.

The applicability of all these results is necessarily limited. Specific utility-sector policies for CO<sub>2</sub> reduction may have different effects in different geographic areas than the effects assumed in these estimates, including changes in the utilization of other technologies besides coal-fired plants. For example, an energy efficiency policy could reduce use of natural gas as well as use of coal. Moreover, policies affecting other sectors - notably transportation - could also generate nontrivial ancillary environmental benefits. Further, health effects do not exhaust all the environmental benefits. Finally, benefits would be larger with non-marginal GHG mitigation policies that drive SO<sub>2</sub> emissions below the regulatory cap.

In light of these limitations, it is tempting to embrace the last three, economy-wide studies in Table 2 that attempt to describe the effects of non-marginal GHG reductions and include a variety of pollutants and impacts. However, the methodologies in these studies simply compute a total economic benefit from a national reduction in criteria pollutant emissions. They lack attention to locational differences in emissions and exposures, and they inherently overestimate the total ancillary benefits from SO<sub>2</sub> reduction by failing to take into account the effect of the SO<sub>2</sub> cap. Hence, they may be better suited for examining the effect of more substantial and broad scale GHG mitigation policies than for examining the effect of more modest policies.

Our analysis using RFF's HAIKU/TAF framework (which underlies the first row in Table 2) leads us to conclude that at least for relatively modest GHG control levels, ancillary benefits may be a significant fraction of costs. The marginal costs of small initial reductions are likely to be fairly low; indeed there is reason to think they would be close to zero (some would even argue less than zero, though we remain skeptical of this). As compared to such a low cost, ancillary environmental benefits of even \$3 per ton of carbon reduced, let alone \$8/ton, could have a significant effect on the volume of no-regrets emissions reduction, especially for moderate carbon taxes of around \$2 per ton. Under such a marginal tax, the average cost per ton of carbon reduced will be less than \$25 per ton and hence ancillary benefits may come close to justifying the moderate carbon tax.

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<sup>142</sup> There are some estimates related to the social costs of transportation. See Greene et al. (1997).



## 6.2 *Lessons for policy*

Some lessons for the design of policy can be derived from our analysis, though they must be interpreted with care. Ancillary benefits may be larger for GHG policies that target coal use, but this has at least as much to do with the continued use of old, relatively polluting boilers as with the use of coal itself. And GHG abatement policies that have relatively greater effects and impose greater costs on newer plants will have the perverse effect of creating a new bias against construction of new facilities, resulting in continued use of older facilities and lower ancillary benefits. By the same token, energy efficiency programs whose effects displace gas use as well as coal will have smaller ancillary benefits.

A second set of lessons concerns spatial differentiation in ancillary benefits. GHG mitigation that occurs in areas especially conducive to the formation of secondary pollutants (ozone and secondary PM), and at sources whose effluent reaches large populations, confer larger ancillary benefits compared to other options.

The possible trend in ancillary benefits over time also is of interest. It is often argued that abatement costs associated with a goal like GHG emissions stabilization will rise over time because of growing energy demand, though this trend will be ameliorated by technical progress and ultimately by a transition to non-carbon backstop energy resources. While this argument is reasonable, one might also expect upward pressure on ancillary benefits per ton of GHGs. This is because of growth in population density and congestion, as well as growth in income, can be expected to yield an increase in the willingness to pay for environmental protection (Krutilla, 1967). On the other hand, improvements in air quality over time will lower the ancillary effects that could be obtained by a GHG policy. There is no way to reach a resolution of these conflicting forces without further analysis.

Cost estimates of GHG policies often fail to anticipate a changing regulatory baseline that is expected to lead to air quality improvements over time and raise the cost of more heavily polluting fuels. Such GHG cost estimates would overstate the relative opportunity cost of GHG policies. In comparing benefits and costs, it would be misleading to include improvements in baseline air quality in calculating ancillary benefits while not including the effect these changes have on the opportunity cost of GHG policies. We correct for this in some of the studies we review in Table 2 by adding in the benefits of avoided investments in SO<sub>2</sub> abatement under the cap that would result from GHG policies.

It is important to be cautious about the implications of ancillary benefits for the desired level of GHG control. Ancillary benefits clearly are important enough that they should be considered jointly with costs of carbon reduction to identify the preferred policies for society. However, the policies that maximize net benefits for society may not be ones that maximize ancillary benefits nor ones that achieve GHG reductions at the lowest gross cost. For instance, a GHG emissions trading program may minimize the direct cost of abatement associated with a GHG reduction target, but it will not necessarily minimize the social cost including ancillary benefits. The preferred policy for achieving a stated level of emission reduction is the one with the lowest net costs of GHG control after allowing for ancillary benefits.<sup>143</sup> An ideal policy would force emitters to recognize the social opportunity costs of GHG emissions together with the costs of criteria air pollutant emissions. At the same time, the choice of policies can have important distributional effects, both in economic costs and ancillary benefits, which must be considered as well.

### **6.3      *Lessons for methodology and needs for further research***

The apparent systematic difference between the estimates achieved in more aggregate models and those in detailed sector specific models suggests an important lesson for further research. The more detailed models provide a fuller characterization of many variables that emerge as important. Among these is behavioral response and detailed characterization of the baseline, both particularly important features of policy analysis.

The virtue of detailed modeling also applies to underlying issues of technological change and demographics. We have noted that changes in population will yield changes in willingness to pay and in the number of people benefiting from environmental improvement. But it is also the case that the location of the population is important to exposure, and demographic trends in the US have implications for how many individuals are exposed to criteria pollutants from electricity generation, as well as their age, an important variable in the concentration-response calculus.

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<sup>143</sup> Our advice presumes that policy will be shaped taking an emission reduction goal as given, or that such a goal will be developed independent of estimates of the direct benefits of GHG reductions. The preferred approach would be to combine ancillary benefits with direct benefits for comparison with costs. One reason is that when considering uncertainty in policy design (Weitzman, 1974), the measure of costs should reflect behavioral responses. To reduce estimates of cost by including ancillary benefits in the cost function would understate behavioral responses, since those responses in reality would be based on costs born privately in compliance with the program independent of ancillary social benefits. Hence, to identify a preferred emission target, a quantitative benefit estimate is implicit, and ancillary benefits should be included on this side of the benefit-cost calculus. However, if the emission goal is explicit and fixed, then we advise that ancillary benefits should be considered with costs to find the least net cost means of achieving that goal.

However, aggregate and general equilibrium models also have virtues. One is the consistency they impose among changes in various sectors in the economy, and another is the linkage to capital and labor markets, which can be important for large policy changes. Perhaps the most important, for the estimation of benefits and costs of climate policies, is the interaction of changes in policy with pre-existing policies and taxes. Evidence of the so-called “tax interaction effect” suggests policies that impose additional costs through regulation are much more expensive from a social perspective than is apparent in partial equilibrium (sector) models.<sup>144</sup> Conversely, the ancillary benefits of GHG policy may be larger than is reflected in sector models because of the reduction in the tax interaction costs from other regulatory policies that, as we have noted, should be specified carefully in the baseline. However, none of the general equilibrium studies we review have addressed this issue.

Another important frontier for research is the calculation of ancillary benefits from GHG policies in developing nations. Review of existing estimates for these countries is beyond the scope of our paper.<sup>145</sup> These benefits may be quite a bit more significant relative to the cost of abatement policies than those measured in the US and Europe, because of lower existing levels of pollution control and lower efficiency in energy use in these latter countries. Speaking in general terms, however, existing developing country studies of ancillary benefits are limited in number and generate highly variable conclusions. The estimates are fraught with uncertainty, for several reasons.

Detailed modeling of how emissions disperse in the atmosphere is rarely available, and detailed emission inventories are rare, so studies often have simply applied “unit values” expressing a change in health status resulting from a change in emissions without modeling emissions diffusion, population exposure, and health responses. Even when these intermediate steps are modeled, studies have used relationships from the US and elsewhere that may not be applicable because of other important influences on health status including differences in expected lifetimes and other risk factors. There is no doubt that lots of potential exists for health improvements in developing countries, but continued uncertainty about how GHG restrictions might contribute to this.

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<sup>144</sup> Goulder et al. (1999), Parry et al. (1999).

<sup>145</sup> Efforts to assess these issues are described in Dowlatabadi (1997) and Davis et al. (1997), as well as other papers in this volume and the forthcoming Third Assessment Report of the IPCC.

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## APPENDIX A: SOURCE INFORMATION (GUIDE TO ACRONYMS)

- A. HAIKU/TAF: Burtraw, *et al.* 2000.
- (2) ICF/PREMIERE: Holmes, *et al.* 1995. Results from this study were combined with analysis using the PREMIERE model cited below.
- (3) Dowlatabadi *et al.*/PREMIERE: Dowlatabadi, *et al.* 1993. Results from this study were combined with analysis using the PREMIERE model cited below.
- (4) EXMOD: Rowe, *et al.* 1995.
- (5) Coal/PREMIERE: Palmer and Burtraw. 1997. Bloyd, *et al.* 1996.
- (6) Coal/PREMIERE/RIA - same as above, plus: U.S. Environmental Protection Agency (USEPA). 1996.
- (7) Abt/Pechan: McCubbin, *et al.* 1999.
- (8) Goulder/Scheraga and Leary: Goulder, 1993. Scheraga and Herrod. 1993. Scheraga and Leary. 1993.
- (9) Boyd *et al.* Boyd, *et al.* 1995.
- (10) Viscusi *et al.* Viscusi, *et al.* 1994.



## **APPENDIX B: DEVELOPMENT OF THE HYBRID ICF/PREMIERE AND DOWLATABADI/PREMIERE ESTIMATES**

To feed the emissions reductions from the ICF study into PREMIERE, we consider the emission reductions for NO<sub>x</sub> that would result from the most influential action studied, Motor Challenge, and estimate health benefits resulting from changes in direct emissions and secondary nitrate concentrations to be \$394 per ton of avoided NO<sub>x</sub> emissions (54,120 tons), totaling \$21.7 million (1996\$). These benefits accrue with a 6.2 million ton reduction in carbon emissions.

The regional percentages of total health benefits that result from these emission reductions vary significantly from the percentages of emission changes themselves. For example, ECAR (the Ohio Valley) produces 19 percent of the emission reductions, but captures 30 percent of the health benefits, due largely to long-range transport from downwind regions to its west. This estimate excludes the contribution of NO<sub>x</sub> to ozone formation, and does not address visibility impairment and other environmental impacts of nitrogen deposition. However, it is likely to capture the lion's share of measurable economic value due to the inclusion of suspected mortality effects, which tend to dominate the economic valuation of conventional pollutant impacts.

To feed the Dowlatabadi *et al.* analysis into PREMIERE, we consider the short run emission reductions for NO<sub>x</sub> that would result from the seasonal gas burn policy. The health benefits that result from direct emissions and secondary nitrate concentrations are estimated by PREMIERE to be \$135 per ton of avoided NO<sub>x</sub> emissions (1.04 million tons), totaling \$141 million (1996\$). These benefits accrue with a 47 million ton reduction in carbon emissions. Note that the benefits per ton are about one-third of the benefits that result from ICF/PREMIERE. This reflects the difference in the location of emission changes in the two models which produces a difference in the atmospheric transport of pollutants and the size of the exposed populations.

### **III. LINKS TO POLICY-MAKING**

# POLICY FRAMEWORKS FOR THE ANCILLARY BENEFITS OF CLIMATE CHANGE POLICIES

by David PEARCE

## 1. The issue

Most environmental policy is targeted at specific goals: a reduction in greenhouse gas emissions, achievement of some interim deposition or concentration level of air pollutants, improvements in water quality etc. These are *policy targets* or *policy levels*. Achieving given targets can involve different policy choices: standard setting, environmental taxes, public information campaigns, negotiated agreements etc. Many of these choices can be combined so that policy measures become *hybrids* or *policy mixes*. Any policy instrument or mix of instruments may have an impact on policy levels which are not the direct target of the policy in question. Thus, climate change policies which involve measures to reduce carbon dioxide emission levels may also have a number of other effects:

- reducing other pollutants that are jointly produced with carbon dioxide, e.g. nitrogen oxides, particulate matter and sulphur oxides;
- reducing other harmful impacts such as (traffic noise, road accidents, and community severance (e.g. loss of neighbourhood due to heavy traffic flows). A policy which seeks to reduce CO<sub>2</sub> emissions by controlling traffic might, for example, have the effect of reducing all these transport-related damages;
- possibly increasing employment levels relative to some baseline in which the climate policy is not adopted;
- possibly stimulating technological change.

Benefits which accrue as a side effect of targeted policies are known as *secondary benefits*, *policy spillover effects*, *'co-benefits'* or *ancillary benefits*. If it is legitimate to credit these benefits to the policy measures in question, then it is clear that their inclusion may change the way in which a policy is viewed. A policy that might not appear to be worthwhile in terms of the benefits and costs of achieving a given policy target, may become worthwhile if the ancillary benefits are credited to the policy. Similarly, some policies may involve the sacrifice of ancillary benefits, so that a secondary cost is involved, perhaps transforming a policy that was worthwhile into one that is not worthwhile. While the focus tends to be on ancillary benefits, the same logic entails that indirect negative consequences of a climate policy should also be accounted for, i.e. there may be *ancillary costs*<sup>146</sup>.

Clearly, knowing the size of ancillary benefits has great potential significance for various aspects of policy and in particular: (a) whether or not *any* policy action is worthwhile, and (b) whether or not the total benefits of a policy can be increased by adjusting *policy design* (see Annex 1).

The issue is perhaps most important in the context of climate policy, but it is not confined to that concern. The reasons that ancillary benefits matter in climate policy are:

- a) that there is some evidence to suggest they could be substantial, thus altering explicit or implicit benefit-cost ratios of emission control policies (Pearce *et al.*, 1996);
- b) related to (a) above: that greenhouse gas control policies tend to have significant economic effects, reflecting the fact that carbon (in particular) is pervasive to the workings of most economies. This pervasiveness of policy effects has made some governments reluctant to embark on greenhouse gas emission reduction programmes for fear of widespread economic costs. The existence of ancillary benefits could make such programmes more attractive by effectively 'internalising' some of the costs of participating in an international agreement (Tol *et al.*, 1995; OECD, 1999);
- c) that policies of carbon trading under the Kyoto 'flexibility mechanisms' will tend to have *ancillary costs* to the investing (permit buying) country because of the reduction in domestic emission reduction action that occurs relative to the case where there are no carbon trades. This may affect the desirability of trading as a means of reducing compliance costs with the Kyoto Protocol (Lutter and Shogren, 1999). However, spillover effects from the investment in the permit-selling country could be positive, so that the *global net effect* could be positive or negative;

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Note that the 'with/without' principle in cost-benefit analysis automatically leads the analysts to account for all costs and benefits that occur because of the policy relative to the baseline of no policy. Nonetheless, ancillary benefits analyses have often adopted different scopes for what is and what is not included as cost and benefit.

- d) that, whereas the *locational source* of greenhouse gas emissions does not matter if the focus is on greenhouse gas damage only<sup>147</sup>, the location does matter if ancillary benefits are accounted for. Such considerations could, in principle, affect burden sharing rules for dealing with greenhouse gas emission reductions, i.e. the geographical distribution of emission reductions. Within the European Union, for example, there is a greenhouse gas emission reduction agreement which shares out the emission reductions required for the EU under the Kyoto Protocol. This burden sharing agreement appears not to have taken account of the associated effects of greenhouse gas reductions on the emissions of transboundary pollutants, despite the fact that there are several transboundary agreements on conventional air pollutants in the EU and in the wider Europe (Pearce, 1992; Heintz and Tol, 1996).
- e) since different policies with equal greenhouse gas emission reduction effects can have widely varying ancillary impacts, policy design and selection becomes more complex.

The issues arising are therefore:

- a) how can the significance of ancillary effects be demonstrated? This is the issue of methodology and demonstration;
- b) once demonstrated, how can ancillary effects be integrated into decision-making? This is policy integration and overcoming barriers to implementation. Additionally, what methodologies are available for incorporating ancillary effects into policy design?
- c) what are the implications of ancillary effects for choice of policy instrument – taxes, standard setting etc.? This is the issue of policy design.

The focus of this paper is on (b) and (c). The first issue is touched on only in so far as it is used to substantiate statements made in this paper.

## **2. Methodologies and demonstration**

There are several reviews of the ancillary benefits literature (Ekins, 1996; Burtraw and Toman, 1997; Burtraw et al 1999; OECD, 1999). The literature tends to focus on *monetary estimates* of ancillary pollution damage from greenhouse gas emissions, the monetary value of reduced pollution damage from climate control, and employment impacts.

### **2.1 *The methodologies used***

Methodologies used in the literature for estimating ancillary effects vary.

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This is because GHGs are *uniformly mixed pollutants*. Each tonne emitted does the same marginal warming damage regardless of its source of emission.

Some of the early literature (Pearce, 1992; Barker, 1993) take a ‘fixed coefficients’ procedure. Emissions of X tonnes of carbon are associated with Y tonnes of some other pollutant, so the value of ancillary damage is  $\$V.Y/X$ , where V is the marginal willingness to pay for the associated pollutant expressed per tonne of carbon. Such coefficients are essentially average and not marginal values, i.e. no account is taken of the future or ‘in the pipeline’ policy context. Marginal emissions could differ significantly from average coefficients and this could explain some of the relatively high values found in the European studies.

Estimates of the money value V in ancillary effect studies typically come from *benefits transfer* studies. Benefits transfer involves taking either a willingness to pay (WTP) value or willingness to pay function from a context where a study has been carried out (the ‘study site’) and applying it to another context (the ‘policy site’). Thus, a mean willingness to pay of \$A for avoiding the damage associated with one tonne of sulphur oxides emissions impact might be applied to the policy context in which Y tonnes of SO<sub>x</sub> are reduced as a result of a climate control policy. Alternatively, some WTP function, e.g.  $WTP = aI + bAGE + cEDUC$ , might be ‘borrowed’ from the study site and applied to the policy site. Here I = per capita income, AGE is average age of the affected population and EDUC = educational attainment of the affected population. The coefficients a,b,c are assumed to take the same value at the policy site, but the relevant values for I, AGE and EDUC are inserted in order to estimate a modified WTP. More detail on benefits transfer is given in Annex 2 which gives an overview of the general problems associated with methodologies using monetary values.

The policy approach simulates the effect of some policy, say a carbon tax, and estimates the reduction in associated pollutants, Y’ for a given policy that reduces carbon emissions by X’. Simulation involves a model of some kind, ranging from some environmentally modified input-output approach to full general equilibrium models. Again, the value of this reduction is then  $\$V.Y’/X’$ . In this case, however, Y’ need not be the same as Y since the reduction in Y is policy-dependent and, furthermore, Y’ may allow for general equilibrium effects whereas the simplistic approach using Y does not. Nonetheless, the ratio Y/X and Y’/X’ remain useful if inexact comparators (see Table 1 below).

Other approaches tend to focus on the physical effects without monetisation of those effects. The unit value V in the monetary approach, for example, should reflect the economic impacts of the associated pollutants on crops, ecosystems, human health and materials damage (and perhaps also visibility). Thus V subsumes a set of dose-response functions relating the pollutants to the various impacts. In non-monetary approaches, the physical effects are highlighted rather than having them valued in monetary terms. Thus, pollution reduction Y or Y’ is linked by dose-response functions to health effects H, say lives saved or life-years saved. An indicator of ancillary benefits is then H or H/X.

## 2.2 *Monetary values*

The monetary results are presented in different ways. Ancillary benefits are usually presented in absolute terms (e.g. \$ per tC), as a multiple of ‘primary’ benefits (i.e. global warming damage avoided), or as a percentage of abatement costs. The focus on ancillary benefits as a multiplier of primary benefits tends to reflect a concern with benefit-cost. The focus on recovery of abatement costs tends to reflect a concern with no regrets policies, i.e. the larger the recovery fraction, the less ‘regret’ there is in climate change policy. OECD (1999) notes that the range of values, expressed per tC, is \$3-88 for the studies with estimates for the USA (ignoring the early studies of Ayres and Walter, 1991) and \$44-305 tC for the European studies<sup>148</sup>.

The earliest references to the potential significance of ancillary benefits in GHG control appear to be Glomsrod (1990), restated in Alfsen *et al.* (1992), Ayres and Walter (1991) and Pearce (1992). Pearce’s 1992 analysis suggested that ancillary benefits might be 8-21 times the ‘primary’ benefits (global warming avoided) in 2010 in the UK and 9-24 times in Norway for the same year, i.e. the UK and Norwegian estimates were similar. Barker (1993) confirmed that UK ancillary benefits were high, using Pearce’s estimates, but building them into a full macroeconomic model.

These early studies formed the basis of IPCC’s assessment of ancillary benefits (Pearce *et al.*, 1996). IPCC was careful to point to the policy context involving ancillary benefits by saying that their existence did not *necessarily* amplify the justification for GHG abatement policy since policy that addressed the ancillary damages directly might still be preferred. It is possible that this view – that ancillary benefits are not relevant to climate policy because those benefits are more cost-effectively secured by policies directly aimed at the relevant pollutants - has inhibited the integration of ancillary benefits into climate change policy analysis, but it is difficult to find evidence that this is the case.

The IPCC study did not suggest any ‘multiplier’ for ancillary benefits, but did note that some studies suggested they could offset between 30% and 100% of abatement costs (Pearce *et al.*, 1996, p.218).

With the exception of Lutter and Shogren (1999), US studies have typically found multipliers of less than unity. A caveat to this conclusion is that it is difficult to secure a ‘normalised’ basis for the comparison, i.e. some studies look at the ancillary damage done from, say air pollutants, associated with the emission of one tonne of carbon. Others simulate the effects of carbon taxes. Studies also differ in scope, some focusing on the electricity sector alone and others estimating economy-wide impacts. The Burtraw *et al.* (1999) study finds ancillary benefits to be almost trivial relative to primary benefit estimates at \$3 tC, but this is for a simulated \$10 tC carbon tax. By contrast, the Lutter-Shogren study finds a value of around \$300 tC.

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Ayres and Walter (1991) is one of the first studies to investigate primary and secondary benefits, but the study has been severely criticised for double-counting warming damages, and hence the benefits of control – see Fankhauser (1994).

It is worth noting that the Global Environment Facility (GEF) has routinely had to address the issue of ancillary benefits in its efforts to give substance and meaning to the notion of 'incremental cost'. GEF grants directed at global warming control have their main justification in the reductions in global GHG emissions. But ancillary benefits mean that recipients of grants may actually secure domestic benefits (e.g. technology transfer, local pollution control, employment) since they do not pay themselves for the GHG control measures. Clearly, a grant policy that ignored these domestic gains would have less effect on global warming control than one that required recipients to pay at least part of the cost of the GHG control policy. In the limit, recipients could pay up to the level of the domestic benefits received and still have an incentive to adopt the GHG control measure. This would free resources for new grants to other recipients, thus expanding the number of project that could be financed with a given budget. In practice, GEF tends to operate so that only some domestic ancillary benefits are deducted from the 'gross' incremental cost. Local environmental benefits may be computed but not deducted. Rosebrock (1994) suggests that such local environmental benefits may have money values of a size comparable to the GEF grant and to global benefits.

### **2.3 *Overview of monetary estimates of emissions-related ancillary benefits***

Table 1 lists estimates of ancillary benefits as a multiple of primary (marginal) benefits. It is important to understand what the computations show since, as noted earlier, the methodologies vary. To repeat the methodology - the ancillary benefits take the form of either emissions of air pollutants associated with the emission of one tonne of carbon, or the reduction in air pollutants associated with a particular policy that reduces carbon emissions by one tonne. The former approach is analogous to a 'fixed coefficients' procedure. Emissions of X tonnes of carbon are associated with Y tonnes of some other pollutant, so the value of ancillary damage is  $\$V.Y/X$ , where V is the marginal willingness to pay for the associated pollutant. The policy approach simulates the effect of some policy, say a carbon tax, and estimates the reduction in associated pollutants, Y' for a given policy that reduces carbon emissions by X'. Again, the value of this reduction is then  $\$V.Y'/X'$ . In this case, Y' need not be the same as Y since the reduction in Y is policy-dependent. Nonetheless, the ratio Y/X and Y'/X' remain valid comparators. Table 1 shows these values in terms of  $\$V.Y/\$V.X$  and  $\$V.Y'/\$V.X$ . While the ratios cannot be strictly compared they give an approximate guide.



Table 1. Ancillary (emission) benefits per tonne carbon as a multiple of primary benefits

Study	Country	Ancillary benefits in \$tC	Ancillary benefits as a multiple of primary benefits (at \$45 tC) <sup>1</sup>	Comment
Ayres and Walter 1991	USA	165	3.67	
Barker, 1993	USA	251	5.58	VOCs
Boyd <i>et al.</i> 1995	USA	40	0.89	Criteria pollutants
Burtraw and Toman, 1998	USA	<10	<0.22	Judgmental assessment of prior studies
Burtraw <i>et al.</i> 1999	USA	3 <sup>2</sup>	0.07	SO <sub>2</sub> , NO <sub>x</sub> only
Dowlatabadi <i>et al.</i> 1993	USA	3	0.07	SO <sub>2</sub> , NO <sub>x</sub> , PM
Goulder 1993; Scheraga and Leary 1993	USA	33	0.73	SO <sub>2</sub> , NO <sub>x</sub> , PM, Pb, CO, VOCs
Lutter and Shogren, 1999	USA	300	6.67	See text
Rowe <i>et al.</i> 1995	USA	24	0.53	SO <sub>2</sub> , NO <sub>x</sub> , PM
Viscusi <i>et al.</i> 1994	USA	88	1.95	Criteria pollutants
Barker 1993	UK	44-201	0.98-4.46	Relies on Pearce 1992
Pearce 1992	UK	195	4.33	SO <sub>2</sub> , NO <sub>x</sub> , PM
Ayres and Walter	Germany	312	6.93	
Alfsen 1992	Norway	102-146	2.27-3.24	
RIVM et al 2000	European Union	53- 79	1.17-1.75	General equilibrium model

Sources: OECD (1999); Lutter and Shogren (1999); RIVM et al (2000). The primary benefit figure for warming damage avoided is from Eyre et al. (1997) – see Annex 2.

Notes:

1. The \$45 damage figure is an estimate of marginal warming damage from the release of one more tonne of carbon now. The damage figure rises over time in some models but only at a modest rate (Pearce et al, 1996).
2. This figure would be approximately doubled if savings on SO<sub>2</sub> control costs are included.

Table 1 reveals a very wide range of estimates. Differences would appear to be due to a number of factors. First, methodologies differ. Some of the estimates are simplistic in that they come from correlations between emissions of CO<sub>2</sub> and the conventional pollutants. These cross coefficients are averages in some cases (e.g. the early UK work) and marginal figures in other cases. Either way they take no account of ‘policy in the pipeline’. The fairly stringent controls on SO<sub>x</sub> and NO<sub>x</sub>, say, would have the effect of lowering the amount of these pollutants emitted, so the cross coefficient will fall over time. Hence these early estimates are almost certainly exaggerations. Other estimates come from the running of various models of different degrees of sophistication. These are more reliable. Second, the estimates derive in some cases from simulations of carbon taxes and the tax rates used vary from a few dollars per tC to very high rates. Since the results in Table 1 are shown in the form of emissions per tonne of carbon, this normalisation procedure should remove most of the variability due to different carbon tax rates, but if higher tax rates have a bigger proportional effect on carbon emissions, then the results will vary.

Nonetheless, even accounting for these differences, the impression remains that ancillary benefits *could* be comparable in size to the ‘primary’ (global warming) benefits. Just as significantly, ancillary benefits offer significant potential for no regrets policies, i.e. significant fractions of abatement costs are recovered, at least for early measures.

## 2.4 Overview of non-monetary ancillary benefits

### 2.4.1 Emissions

As an example of ancillary emissions reductions that are not monetised, Bernow and Duckworth (1998) estimate that a policy package in the USA which reduces CO<sub>2</sub> emissions by 10% by 2010 (relative to 1990) would have the following effects: a reduction in SO<sub>2</sub> emissions by 5.5 mt SO<sub>2</sub> over and above Clean Air Act achievements compared to the 2010 base case; 4 mt NO<sub>x</sub>; c8 mtCO; 300,000 tPM, and 750,000 tVOCs<sup>149</sup>.

An OECD wide perspective is provided by Complainville and Martins (1994) Using the OECD ‘GREEN’ model they simulate an escalating carbon tax and estimate the effect on CO<sub>2</sub>, SO<sub>x</sub> and NO<sub>x</sub> emissions. Some of the results are shown below.

Table 2. **Effects of a carbon tax on NO<sub>x</sub> and SO<sub>x</sub> emissions: OECD ‘GREEN’ model**

*[Tax of \$22 tC in 200, rising to \$156tC in 2050. Impact shown as % deviation from baseline emissions]*

	2000			2050		
	CO <sub>2</sub>	NO <sub>x</sub>	SO <sub>x</sub>	CO <sub>2</sub>	NO <sub>x</sub>	SO <sub>x</sub>
USA	-8	-8	-13	-55	-45	-64
Japan	-4	-3	-4	-48	-35	-41
EU	-5	-4	-7	-45	-34	-48

Source: Complainville and Martins (1994).

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The implied cross-coefficients are not very different to those derived by Pearce (1992) for the UK, i.e. tonnes SO<sub>x</sub> etc. per tC reduced.

The most notable observation is that, in terms of percentage deviations, the effects on NO<sub>x</sub> and SO<sub>x</sub> are as pronounced as they are for carbon dioxide.

#### 2.4.2 *Health benefits*

An example of ancillary benefits expressed as ‘lives saved’ is given by The (international) Working Group on Public Health and Fossil Fuel Combustion (1997). They estimated that a climate policy that reduced developed country CO<sub>2</sub> emissions 15% below their 1990 level would save a cumulative total of some 8 million lives from 2000 to 2020 due to ancillary reductions in particulate matter. Some 6.3 million of the total would be in developing countries and 1.7 million in developed countries.

#### 2.4.3 *Employment*

Employment effects tend to be estimated in studies that adopt macroeconomic or general equilibrium modelling of climate improvement policies. Most studies find that climate control policies have a net cost in terms of GNP and hence, most likely, a net cost in terms of employment (Hourcade, 1996). A few studies, notably for the UK, suggest employment gains from GHG control. The former might suggest ancillary costs, the latter ancillary benefits.

There are a number of problems associated with including employment as an ancillary effect.

First, in a fully fledged cost-benefit study the net effects of any measure will be measured in terms of changes in human wellbeing. If wellbeing is measured in terms of changes to GNP, then adding in employment gains as well as the GNP gains would be double counting. Properly conducted, CBA would identify wellbeing changes that subsume the GNP gains, so that employment would properly be excluded from any benefits assessment. As we note later, however, ancillary effects analysis may well not take the form of a cost-benefit study. If so, and provided care is taken to ensure that double counting is not present, gains (or losses) in employment could be included.

Second, the extent to which employment changes as a result of GHG policy is sensitive to the form that the policy takes. Most notable is the sensitivity to revenue-recycling, i.e. to the use made of any revenues from carbon taxes and (less likely) auctioned tradable permits. As a general rule, if revenues are recycled, employment gains will be higher. However, there is a substantial debate on which form of recycling matters most – notably reductions in labour taxes, reductions in personal income taxes, reductions in corporate income tax or increases in investment allowances (Shackleton et al, 1992). Some experts cast doubt on whether any of these apparent benefits should be regarded as a credit to GHG control, policies. The reason for this is that the size of the employment/GNP gain is determined by the scale of the economic distortion already embodied in the tax that is reduced with the recycled carbon tax revenues. But if the outcome occurs because the taxes that are reduced are already distortionary, those taxes should be reduced anyway. It therefore amounts to ‘claiming too much’ for climate change policies that they have these employment effects. Against this, one has to ask whether such anti-distortionary policies would have been taken in the absence of GHG control measures. GHG controls will impose some costs on emitters and it could be argued that revenue recycling is away of ‘buying’ the co-operation of emitters. Put another way, the issue is one of what constitutes the ‘political baseline’: a set of distortionary taxes that should have been eliminated (or reduced) anyway, or a set of distortionary taxes that are not likely to be reduced in the absence of climate change policy.

Overall, if the ancillary benefits issue is regarded in purely analytical terms, employment gains – if they occur – would not seem relevant. Either they reflect gains in wellbeing that are already accounted for in, say, a cost-benefit analysis, or their occurrence reflects distortions in the economy rather than the beneficial effects of climate policy. There is, in effect, no ‘double dividend’. A more down-to-earth view would add in employment gains if the decision-guidance is not presented as a cost-benefit analysis, though care has to be taken not to double count. Similarly, the double dividend argument appears more sound if the climate change policy really can be seen as the route for reducing distortions that would not otherwise be reduced

#### 2.4.4 *Technology-forcing*

Arguably, a GHG control policy will stimulate technological change in the abatement sector. However, the extent to which such technological developments should be regarded as an ancillary benefit is debatable. If the benefits of the improved technologies ‘spill over’ to other sectors –e.g. to the control of other pollutants – then it would seem correct to count any likely cost savings as a benefit to climate control measures. If the cost reductions are confined entirely to the GHG control sector then it is less clear that the future cost reductions should be credited to the policy since those cost reductions should be built into the GHG abatement cost estimates. There appears to be very little evidence about the effects of policy measures on technology spillovers, although there are some studies that try to estimate the technology forcing impacts of policy *in general*. Thus, Kemp (1997) finds that impacts on technical change are dependent on the form that policy takes. In contrast to many of the theoretical expectations, he argues that tradable permit systems are the most technology-forcing and environmental taxes the least forcing, with traditional command and control measures in between. The comparatively poor performance of taxes reflects the fact that they tend to be introduced at too low a level for fear of industrial lobbies against them and perceived impacts on competitiveness. Thus it is important to distinguish the likely effects of ‘properly’ designed taxes from the taxes that are likely to be introduced in practice.

Kemp (1997) does consider the more relevant issue (for the current topic) of how far any new technology is *diffused*. GHG control technology will, for example, largely focus on energy saving and energy saving is just as relevant to conventional pollution control. The results of Kemp’s analysis of various case studies in the Netherlands are not clear-cut. Diffusion of conservation technologies is not readily explained by cost advantages. Even the availability of subsidies to energy conservation could not explain take-up. Institutional factors mattered, e.g. the take up of energy conservation in rented homes was as large as in owner-occupied homes, a result mainly due to the environmental consciousness of the non-profit housing councils that own most rented accommodation. But the potential for diffusing technology across different sectors is high because technological solutions often exist outside of the sector being targeted.

Overall, there is some evidence that ‘technology forcing’ is an outcome of the kinds of policies that would be relevant to GHG control. However, the extent to which this forcing effect can be regarded as an extra, ancillary, benefit of GHG control seems best supported if the effects are diffused outside of the sectors targeted for GHG control. This ‘‘partial diffusion’’ of benefits is difficult to identify but appears to exist. Its size is probably not something that can be demonstrated in quantified terms.

## 2.5 *Should ancillary benefits be counted at all?*

Some economists doubt if ancillary benefits should be included at all in an analysis of the benefits and costs of climate change policy. Shogren (1999) warns against the possibility of double counting since existing policies targeted at the ancillary concerns (e.g. acid rain) may well account for future reductions in ancillary pollution. Counting them in again would be to exaggerate the benefits of climate control policies. This risk can be overcome by ensuring that any ancillary emission reductions are truly incremental to ancillary policies. Shogren notes that if ancillary emission policies are effective they will capture the net benefits of controlling ancillary emissions. This is true but it presupposes that ancillary emission policy is itself optimal. Cost-benefit studies of acidification control, at least in Europe, suggest that there are still major net benefits to be captured.

### 3. **Policy integration and overcoming barriers to implementation**

#### 3.1 *Decision-criteria and ancillary benefits: no regrets policies*

Faced with a trade-off context, any policy maker will first try to secure a no-regrets policy stance. There are various meanings to 'no-regrets'. We can distinguish three for purposes of analysis.

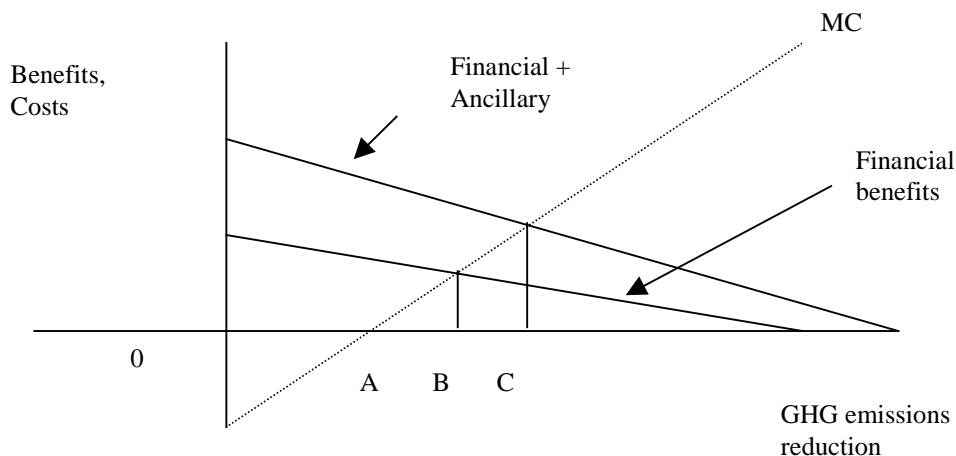
First, no regrets may be defined in such a way that all measures having negative or zero costs are implemented. Since the benefits are assumed to be greater than zero, there is no need to quantify benefits. All measures can be ranked according to the size of negative costs. Strictly, the measures need to be normalised in terms of the tonnes of GHGs reduced, i.e. the appropriate indicator becomes (negative) cost per tonne of Cequ reduced.

Second, no regrets could be defined as a context in which *financial* gains exceed any (positive) financial costs of the policy. One example would be the financial savings from an energy conservation programme which could outweigh the financial costs of the policy. A more subtle example might be that climate policy reduces local pollutants which in turn reduces morbidity and premature mortality. In turn, these benefits are associated with some financial returns, such as reduced health care expenditures. Overall willingness to pay to reduce the health effects would, not, however, be part of the cost-benefit equation since this does not have a direct financial flow associated with it.

Third, no regrets could be defined as a context in which the costs are covered by the financial benefits and monetised estimates of ancillary benefits. Dessus and O'Connor (1999), for example, find that, even with conservative assumptions about willingness to pay for health benefits, Chile could reduce CO<sub>2</sub> emissions by 10% from a 2010 baseline level without any losses to national wellbeing, all costs being offset by ancillary benefits.

Figure 1 illustrates these three different concepts. Benefits and costs in money terms are measured on the vertical axis, and GHG reduction levels on the horizontal axis. MC is the marginal cost of emissions reduction. The line marked 'financial benefits' refers to marginal financial gains. The line marked 'financial and ancillary' refers to marginal aggregate financial and non-financial gains. Note that the global benefits of reduced warming are not included in this analysis. The very narrow definition of no-regrets is illustrated by all actions up to point A. The definition involving financial benefits covering costs is shown by level B, and the definition involving ancillary benefits is shown by level C. As can be seen, assuming there are no ancillary costs, the effect is to increase the warranted level of GHG reduction as the definition is relaxed.

Figure 1. No regrets measures



Some of the climate control literature lends support to the narrow definitions of no-regrets being the case, at least for the early actions on climate change control. This is because mainly ‘bottom up’ cost studies have argued for significant actions that have negative financial costs, especially in the area of energy conservation. If this were true then finding any positive ancillary benefits would be ‘icing on the cake’ – climate policies essentially pay for themselves and the reduced local pollution and other benefits are an incidental gain. The problem is that there are extensive doubts surrounding the view that control costs can be negative since the question is why such investments are not undertaken automatically. Others note that the almost exclusive preoccupation of the business and economics literature is with the *management* of corporations, which suggests that there are issues that need to be managed. What is privately profitable simply does not happen automatically. Hence negative costs may exist but may not translate into automatic action by corporations. There may be many informational, principal-agent and institutional obstacles to implementing good policy<sup>150</sup>. If so, a climate control policy might be as good as instrument as any to draw attention to those negative costs and to initiate managerial policies to exploit them. Indeed, this has been the thrust of many environmental audit and reporting schemes.

On this narrow definition of no-regrets the role played by ancillary benefits is essentially a supportive one, i.e. policy should be self-financing without consideration of ancillary benefits.

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This literature is reasonably large. See, for example, de Canio (1995).

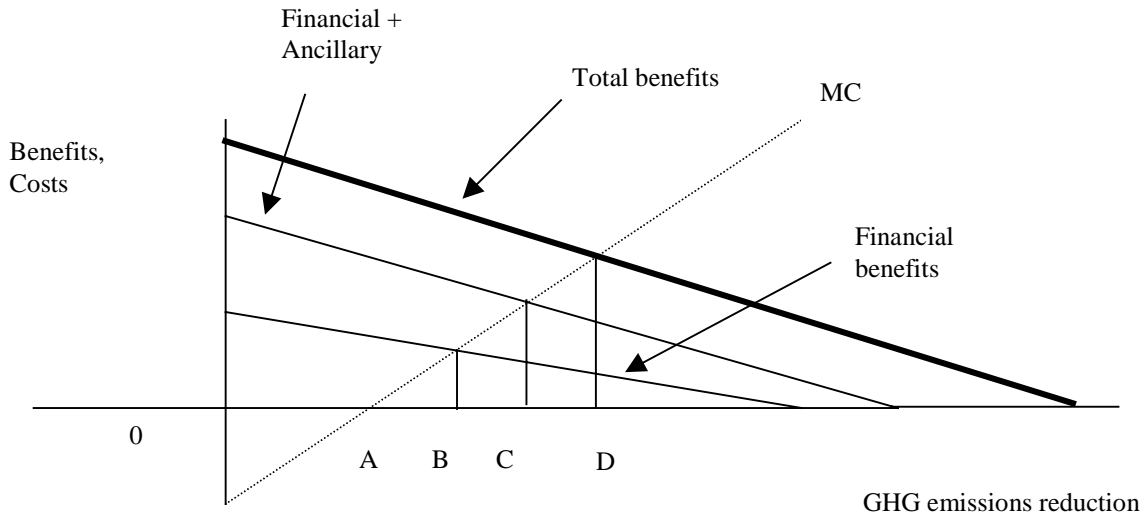
The broader definition of no regrets (option C in Figure 1) suggests that climate control measures may not be financially beneficial in terms of private costs, but are beneficial to the nation implementing controls when ancillary benefits are added. The policy difference is considerable. Self-financing measures centre round information, auditing, exhortation and ‘public reward’ (e.g. prizes). Measures that require their justification in terms of ancillary benefits are far more likely to need financial incentives such as taxes and subsidies. Politically they are obviously far more difficult to secure since there will always be losers. Just as we identify a ‘loss aversion’ culture among political decision-makers (see below), so there is a ‘loser syndrome’ in policy making in general. As long as there are losers who are not clearly compensated for their losses, they have a strong incentive to lobby against the measures in question. Hence, getting ancillary benefits ‘on to the agenda’ is far more difficult than managing the narrow concept of no regrets: for now there are losers and there is no obvious way in which the gainers (from reduced pollution, say) can compensate the losers (the firms bearing the costs).

### **3.2      *Decision-criteria and ancillary benefits: cost benefit analysis***

If cost-benefit analysis (CBA) was the sole procedure whereby decisions about environmental policy were made, then, apart from the issue of reliability of estimates, the ancillary benefits issue would not be so controversial. The reason for this is simple: CBA requires that all costs and benefits be accounted for when computing a cost-benefit ratio or present value total. That accounting procedure operates on the ‘with/without’ principle, i.e. costs with the policy are defined relative to the baseline of ‘no action’. Similarly with benefits. Hence, in a ‘cost-benefit world’, the issue of how to get ancillary benefits into practical decision making would be redundant. It would tend to happen automatically. One technical qualification to this view is that benefits that take the form of stimulated technical change or employment might not easily be ‘monetised’. Hence one could have a situation in which monetary benefits are less than monetary costs but employment and technical change benefits are positive.

The cost-benefit paradigm is illustrated in Figure 2 which repeats Figure 1 but now with the global benefits added. The global benefits take the form of reduced global warming damage. Chapter 2 suggested that ‘benchmark’ figures for (marginal) damage might be around \$45 tC. Note that the cost-benefit approach is here interpreted to be inclusive of global and domestic benefits, even though the benefits of action do not accrue to the emissions-reducing nation. The rationale here, of course, is that the very purpose of emissions reduction is to secure global benefits so that it is legitimate to account for the benefits of reduced warming, regardless of to whom they accrue. Note that the ‘optimal’ level of emissions reduction is now at D.

Figure 2. Cost-benefit approach



While cost-benefit analysis is widely used in OECD countries it is not used consistently nor, when it is used, is it necessarily carried out with strict adherence to the ‘with/without’ principle. More relevant is its limited use in the context of climate change policy.

Since all countries with agreed targets under the Kyoto Protocol negotiated those targets at Kyoto there is no suggestion that formal CBA determined the burden-sharing targets, but the question can be asked as to whether existing CBA studies influenced the negotiating stance of the countries in question. It seems likely that various cost-benefit studies had some influence. Studies such as Nordhaus (1991, 1994) and Manne and Richels (1992) had suggested either that the ‘optimal’ trajectory of carbon emissions differed very little from ‘business as usual’ trajectories, or that significant control was far better undertaken well into the future rather than currently. Nordhaus’s 1994 study suggested that even the voluntary emissions ‘freeze’ agreed at Rio in 1992 had negative net benefits<sup>151</sup>. There is little or no evidence to suggest that these studies were very influential, however. As many commentators have noted, policy-makers have been far more concerned with the short-term costs of mitigation than with longer-term balances of benefits and costs. The IPCC review of abatement costs found substantial disparities between ‘bottom up’ studies – suggesting negligible or small unit control costs – and ‘top down’ studies which suggested annual costs equivalent to 1-2% of GNP (Hourcade et al, 1996). From a policy-maker’s standpoint, such ranges are not treated as if they have a simple expected (mean) value because there is substantial ‘loss aversion’ in decision-making.

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These studies appear to have had more influence than those claiming high benefit-cost ratios – e.g. Cline (1992) – perhaps because the more action-oriented studies still suggested costs in excess of benefits for the first one hundred years. See for example, the graph in Cline (1992, p280).



Far more weight will be attached to the adverse estimates (i.e. the high abatement costs) than to the estimates suggesting only a small cost sacrifice. Loss aversion is easily explained. The perception is that the gains from climate change control will be long-term, whereas the costs of control will be immediate. This perception is correct even allowing for the strong possibility that climate impacts from global warming are already being felt. Even if strong action was taken today, there would be no discernible effect on rates of warming for considerable periods of time due to thermal lag effects. Hence anyone advising that strong action needs to be taken now would in effect be arguing for potentially significant costs to be incurred now for no identifiable benefit over fairly long periods. The role that ancillary benefits could play here is noteworthy since the policy lag effects for other pollutants are substantially less. Thus, while global warming benefits are long term, ancillary benefits could be fairly short-term.

In so far as ancillary benefits are presented in monetised form, it seems fair to say that their integration into climate policy decisions is problematic for precisely the same reason that cost-benefit is problematic. While it is far from clear that any other form of decision-guidance performs better, it remains the case that cost-benefit is controversial.

Widely used for regulatory impact appraisal in the USA, the impact of cost-benefit analysis elsewhere has been limited. The European Commission now regularly subjects planned Directives to cost-benefit appraisals, and there is strong support for cost-benefit in the UK and Scandinavia. Other countries are known to experiment with cost-benefit analysis, but most decision-making is only partially informed by quantitative techniques generally, whether cost-benefit or some other technique. On the various country experiences, see Department of the Environment (1997), Pearce (1998a) and EFTEC (1998) for the UK; Navrud and Pruckner (1997) and Pearce (1998b) for the European Union; Hahn (1996), Morgenstern (1997) and Farrow and Toman (1999) for the USA; and Nyborg (1996) for Norway.

The variable use of cost-benefit analysis can be explained by a number of factors, some of which, it is important to note apply to any quantitative procedure.

First, in so far as the techniques are on the political agenda, the controversies listed in Annex 2 have served to limit its credibility. Some issues, such as the appropriate value for a 'statistical life', are the subject of continuing debate among academics. The sensitivity of so many cost-benefit studies to this value suggests that it is an issue in need of urgent resolution before cost-benefit is likely to secure a stronger foothold in the decision-making practices of many countries (Pearce, 1998a, 1998b). Similarly, ancillary effects appraisal to date has been exclusively dependent on 'benefits transfer' estimates. Yet it is only recently that any significant effort has been made to test the validity of benefits transfer. While some exercises, whereby transferred values are compared to values derive from original valuation studies at the same site, find 'acceptable' errors in the transfer process (Ready et al, 2000), others suggest that transferred values are nowhere near to the values derived from original studies (Barton, 2000).

Second, the institutional structure of the decision-making units within government militates against fully integrated policy making (this holds whether the benefits are monetised or not). Thus the divisions of government making decisions about policy stances on climate change may be divorced from those making decisions about local or regional air quality. Repeated efforts to 'join up' government departments have not been very successful in many countries, so that policy thinking tends to be dictated by single rather than multiple goals. Perhaps just as seriously, climate decisions may not be informed by economic thinking over and above the costs of abatement since climate is regarded as an issue of 'science' first and the politics of international negotiation second. The cost-benefit paradigm may have little relevance to this institutional structure.

Third, whilst thinking about ancillary benefits does not have to involve cost-benefit analysis, there is evidence to suggest that almost any formal guidance procedure that is quantitative in nature is resisted by politicians. Nyborg (1996) interviewed Norwegian politicians to test their reactions to CBA and found that, at best, benefit-cost ratios were used to 'screen' options, whereas the real process of ranking options was regarded as a 'political' process which was sensitive to whatever local conflicts there were. EFTEC (1998) interviewed UK civil servants and found a number regarded techniques such as CBA as inhibiting of the flexibility that politicians preferred to maintain. That is, CBA tended to remove the element of choice that politicians wished to have. Note that this objection may not hold so strongly with other techniques such as multi-criteria analysis where there is more scope for the decision-makers themselves to adopt the relevant scores and weights. In CBA there is no independent scoring of impacts, and weights are set by estimates of WTP. Navrud and Prucker (1997) found that European decision-making 'culture' was not so oriented towards economic efficiency as in the USA. Hence techniques such as CBA are downplayed. Pearce (1998a) found that CBA was more widely used in the European Commission because of revisions to the Treaty of Rome requiring that some form of cost-benefit balance be applied. It was not obvious, however, that the studies had been influential in modifying Directives.

Fourth, like any quantitative technique, CBA requires some degree of technical understanding. Its influence is therefore limited to those with an economics training or those able to apply themselves to acquiring the essentials of the subject. Since it is not a static procedure – techniques change fairly rapidly – there are serious learning costs involved. This may inhibit the use of the techniques.

How far, then, would these obstacles to implementing ancillary benefits analysis be overcome if non-monetised impacts were measured instead? Certainly, issues of technical understanding and decision-maker flexibility would be overcome to a considerable extent. It is also possible that the scientifically trained would be more 'at home' with non-monetised approaches. But such approaches would do nothing to avoid the institutional separation of climate change policy from policy concerned with more localised or national benefits. Nor would some of the technical controversies in CBA be avoided – e.g. the level of population over which to aggregate the non-monetary benefits. Finally, while it is often suggested that monetised costs and benefits add to uncertainty in decision-making, the uncertainty associated with the alternative is often overlooked. It is not clear, for example, that the uncertainty of decision-making associated with monetised estimates is any less than the uncertainty associated with not having any monetised estimates at all.

All this suggests that the search for a refined methodology of ancillary benefits analysis can contribute something to raising the profile of ancillary benefits in policy making, but that some of the obstacles are more deeply seated in institutions, administrative culture and technical capability.

A major emerging area where benefit-cost considerations are relevant, however, is in the determination of the stance that individual countries are taking with respect to carbon trading under the various Kyoto flexibility mechanisms. The essence of the situation is that trading shifts the control effort 'offshore' to EITs (in the case of joint implementation) and developing countries (in the case of the Clean Development Mechanism). But if carbon is reduced offshore none of the joint benefits of reducing carbon 'onshore' is realised. There is little evidence that this domestic cost of trading affected national attitudes at Kyoto, and the issue was not really relevant prior to Kyoto since, although there have been hundreds of actual or simulated trade, their focus has been on learning to execute, monitor and verify such trades, rather than on their costs and benefits. It thus appears to be very much a post-Kyoto concern. Attitudes in the European Union have also only recently begun to take account of the loss of ancillary benefits (it is – January 2000 – an active issue under consideration). Most considerations of trading have centred on the essentially moral-cum-political issue of whether trading reduces developed countries' 'responsibility' for GHG emissions<sup>152</sup>. Recent studies, however, have raised the profile of trading by noting (a) the substantial reductions in costs achievable through trading, and (b) the loss of significant ancillary benefits if trading takes place (e.g. RIVM *et al.*, 2000). Since the physical 'metric' consists of tonnes of different gases (CO<sub>2</sub>, NO<sub>x</sub> etc) the trade-off between cost reductions and loss of ancillary benefits can only be expressed in monetary terms. Hence the new interest in the money value of ancillary benefits.

Overall, then, monetised ancillary benefits have, until very recently, played virtually no role in determining decision-maker attitudes to the policy mix of control measures. There is now evidence to suggest that such estimates are regarded as being relevant to one significant part of GHG policy, namely the extent to which carbon trading should be 'allowed' and encouraged.

### **3.3 *Decision-criteria and ancillary benefits: risk assessment***

The term 'risk assessment' (RA) is used in different ways within the OECD countries. The main differences are:

1. looking at the risks (probabilities and size of hazard) without monetising the risks and without considering the costs of reducing risks;
2. looking at the non-monetised risks and comparing them to the costs of control (cost-effectiveness);
3. looking at monetised risks and the costs of control (cost-benefit analysis).

Other variants exist (see EFTEC, 1999) but these are the main ones of concern. Since cost-benefit analysis has been discussed, only a) and b) are relevant to this section.

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There is an additional concern. The less abatement that occurs 'at home' the lower might the stimulus be to technological change which lowers future abatement costs for greenhouse gas emissions. Thus more trading could effectively keep abatement costs higher than they otherwise would be. I am grateful to Gene McGlynn for drawing this to my attention.

Ancillary benefits analysis fits neatly into the framework of risk assessment. As with CBA, the aim would be to identify the benefits of any global warming strategy, where the benefits will be GHG emission reductions, ancillary pollutant emission changes, and perhaps any technology-forcing effects and employment effects. Risk assessment that ignores costs altogether is not economically rational (although it does describe how some chemical RAs are carried out). Hence, focusing on cost-effectiveness, the obvious issue is how to deal with multiple benefits and a single cost figure. So long as there is ‘vector inequality’ between different policy options, the problem of multiple effects does not matter<sup>153</sup>. But once benefits and costs vary between policy options, some form of weighting is required. CBA presents one form of weighting using (marginal) willingness to pay as the weights. Multicriteria analysis is another option. In both cases there are serious problems of comparability. Long term effects (such as warming reduction) and being traded off against short-term effects (such as reduced traffic congestion etc). Some people adopt a standpoint that such trade-offs are not ethically acceptable since the current generation will automatically ‘vote’ to reduce short-term impacts at the expense of long term impacts they are unlikely to bear. This issue defines an on-going debate in the global warming literature, with some commentators taking the view that it is just as indefensible to allocate resources now for long-term gains to generations who are likely to be better off anyway, at the expense of the underprivileged now (Schelling, 1992, 1999).

The central point is that ancillary benefits can be integrated into ‘standard’ decision-making methodologies, but that in moving away from ‘single target’ policies aimed at reducing global warming there is inevitably a trade-off between some effects and climate effects. Put another way, the chances that climate policy is a ‘win win’ positive sum game are not high. This contrasts with some of the more radical claims made in the climate policy literature.

### **3.4      *Decision-criteria and ancillary benefits: rapid appraisal***

While in principle ancillary effects analysis can be accommodated fairly straightforwardly into no regrets, CBA and risk analysis, there is a demand among decision-makers for less sophisticated but more rapid assessment procedures. Some of this demand reflects limited time and resources for fuller-scale study, but some of it also reflects the current need simply to demonstrate the relevance of the ancillary benefits argument. This process of demonstration is not dependent on numbers being very precise.

The most basic rapid appraisal technique is the *checklist*, whereby when looking at different GHG policies, decision-makers are automatically reminded of the need to consider the ancillary costs and benefits of any chosen action. At the minimum, the checklist simply lists the like ‘cross effects’. Thus a transport-targeted policy would list potential effects on noise nuisance, vehicle pollutants, congestion etc. An electricity-targeted policy would list impacts from other pollutants etc. It is not enough to list the associated emissions, since decision-makers need reminding of the impacts of those emissions. While simple checklists of this kind may seem redundant in a world where there are sophisticated decision guiding procedures available, interviews with numerous decision-makers across a number of countries suggests that their provision is essential.

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<sup>153</sup> Let the benefits of a policy be  $B_1, B_2, B_3$  and  $B_4$  where the benefits are in non-monetary units. Let the cost be  $C$ . Vector inequality exists when  $B_1/C, B_2/C, B_3/C, B_4/C$  is greater than the corresponding ratio of benefits to costs for an alternative policy.

After this simple provision, checklists can be expanded to various levels of sophistication. Thus, ‘cross coefficients’ indicating the likely tonnage of pollutant reduction per tonne of GHG reduction could be provided, sector by sector, policy by policy. Some indication of the importance of impacts could be provided, e.g. by noting that, say, reduction in particulate matter is thought to be the most important with respect to human health effects.

Several problems emerge with rapid appraisal approaches. In principle, these are issues that can be accommodated within the cost-benefit and risk appraisal paradigms. In practice, it is likely that the issues will present problems even for these approaches.

The first issue concerns the timing of the benefits and costs. For climate change the perception is that the costs have to be borne now and the benefits are very long term. For the ancillary effects benefits are shorter term and costs are also borne now<sup>154</sup>. Traditionally, decision-makers have difficulty when dealing with long time horizons and this helps to explain some of the reticence to enter into radical agreements on climate change control. Attempting to integrate concerns with such different time horizons is problematic. The cost-benefit approach ‘solves’ the problem through the practice of discounting, but discounting long term effects tends to conflict with concerns about intergenerational equity, as is well known.

The second issue concerns the scientific certainty about effects. Climate change impacts remain uncertain. The environmental impacts from conventional pollutants are far better known, although considerable uncertainty surrounds some of the effects, e.g. the links between acidic pollutants and forest damage for example. Effectively, then, the decision-maker is being invited to ‘mix’ issues with very different ranges of confidence attached to them. Again, the cost-benefit approach would handle this with various approaches to uncertainty: sensitivity analysis, expected values, expected utility approaches, decision-criteria. While this may still leave the decision-maker with concerns over the validity of integrating ancillary benefits into climate policy, uncertainty is perhaps not as major a challenge as the time dimension issue.

The third issue concerns the availability of technical solutions, many of which, but not all, will be ‘end-of-pipe’ technologies. Conventional pollution tends to be tackled with technology requirements (see Section 4.1.1). While end-of-pipe solutions exist for greenhouse gas emissions they are currently not widely regarded as part of near-term climate change policy packages, mainly due to the need to demonstrate such technologies and their cost. Climate change can therefore only be tackled in the near-future through behavioural change, including fuel switching, energy conservation, and renewable energy. As noted above, that change has to be widespread simply because carbon is pervasive to the modern economy. Again, it may therefore appear to a decision-maker that the two sets of impacts need separate rather than integrated treatment.

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This contrast should not be exaggerated. Ecosystem impacts from acid rain tend to be long term, as is ecosystem recovery when pollution is reduced.

### 3.5 *Conclusion on barriers to integrating ancillary effects into policy analysis*

This chapter has surveyed some of the salient barriers to integrating ancillary effects into policy analysis. Section 3.1 suggested that the presence of ancillary effects could greatly enhance the potential for ‘no regrets’ approaches to climate change policy. In turn, this should increase the political acceptability of climate policies. If there are financial no-regrets, i.e. there are real cash flows accruing back to those who bear the cost of the policy, then corporate opposition to climate policies may be reduced. Where the no regrets extend to non-monetary gains, then governments should, in principle, anyway, also be induced to adopt more rather than less climate control. But non-monetary gains will not necessarily accrue to those bearing the cost of climate policy so that there may still be ‘losers’ in an apparently no-regrets policy. What is no regrets at the social level need not be no regrets at the individual firm or industry level.

Section 3.2 looked at the cost-benefit paradigm and noted that, if properly executed, cost-benefit analysis automatically accounts for ancillary benefits and costs. Moreover, given that governments have positive discount rates, the fact that ancillary benefits occur in the near term (as well as the long term) whereas primary benefits from climate change occur only in the long term means that ancillary effects may further induce governments to adopt climate change policies. The drawback is the uncertainty surrounding monetised estimates of climate change damage. It is far from clear that uncertainty is reduced by omitting monetised damage estimates - a different form of uncertainty is introduced, namely any idea of whether gains exceed costs - but there is a political perception that monetised estimates have low credibility, especially where they include global estimates of non-market effects such as changes in disease vectors, ecosystem change and so on. Section 3.2 also raised an institutional question. Despite the claims of some governments concerning the integration of environmental issues into all government policy, much policy continues to be ‘compartmentalised’ within different agencies and ministries. This inhibits those charged with the responsibility of effecting climate change policies from taking account of ancillary effects since the two policies are unlikely to be integrated within one institution. Perhaps this observation explains why, despite the natural potential of cost-benefit analysis to handle the ancillary effects issue, the cost-benefit approach still seems under utilised.

Section 3.3. argued that ancillary effects are similarly fairly easy to incorporate into risk assessment procedures. The qualification is that one more impact is added to other impacts which are not in monetised form, thus requiring the use of some form of multi-criteria analysis.

If formal appraisal procedures are not widely used, how can ancillary effects analysis be integrated into the less formal ‘rapid appraisal’ approaches to policy? Section 3.4 suggests that, in principle there is no difficulty in adding the ancillary effects into checklists and approaches using ‘scores’ and ‘weights’. In practice the absence of formal rules for selecting impacts may lead ancillary effects to be overlooked, something that is less likely with formal procedures such as cost-benefit analysis where the analyst is trained to look at overall effects. Rapid appraisal techniques are also less likely to accommodate the difference in the timing of costs and benefits. Overall, a guiding rule might be that the less formal the appraisal technique, the more likely it is that ancillary effects will not be fully integrated in policy analysis. Other concerns are that ancillary effects will tend to be local or regional (transboundary) whereas climate change is global. Those trained to analyse local effects may be less likely to exhibit a concern for global impacts. Finally, all formal procedures have the risk of displacing political authority. A good cost-benefit study will produce clear answers which could easily limit the flexibility of politicians to react in the way they wish.

Does the analysis point to any solutions? Probably the most important advance is that the use of formal procedures for policy appraisal is far more likely to account for ancillary effects compared to the use of ad-hoc and rapid appraisal techniques which still appears pervasive to decision-making. The introduction of guidance on regulatory impact appraisal – i.e. the formal appraisal of all regulations, as in the US and, increasingly in European countries such as the UK– would be a major advance.

#### **4. Policy design and ancillary benefits**

We take the design of policy to involve the means or instrument whereby a given target is achieved. It is convenient to divide policy design into three types of choice, acknowledging the reality that policies are increasingly involve hybrid mixtures of instruments:

4. the choice of overall policy stance which we characterise by the usual distinction between ‘command and control’ and ‘market based instruments’,<sup>155</sup> where we take the latter to include the emerging focus on voluntary and negotiated agreements;
5. the choice of instrument within the market based instrument ‘menu’ – e.g. taxes, tradable permits etc.;
6. the choice of mixes of instruments, e.g. taxes and negotiated agreements;
7. the choice between technologies if a command and control approach is adopted.

Probably the single most important feature of the ancillary effects discussion is that environmental policy needs to take far more consideration than it has done of the synergies between policies. Climate change policy, in particular, appears to have significant spillovers to other policy areas. Yet, until very recently, environmental control policies have tended to be focused on single targets. To take the example of the Convention on the Long Range Transport of Air Pollution (LRTAP), while the detailed analysis supporting the various Protocols has involved careful assessment of interactions between pollutants, the actual Protocols have tended to proceed pollutant by pollutant (SO<sub>x</sub>, NO<sub>x</sub>, VOCs). More recently, a multi-pollutant, multi-effect stance has been taken which recognises the interaction between control measures. To some extent, the target-setting procedure does account for climate control policies since the ‘baseline’ against which the target scenarios are compared includes a forecast of energy use. In turn, energy use is partially dependent on climate policies. Nonetheless, the procedures have not so far raised the issue of the ‘best’ mix of climate and pollution control measures.

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The distinction is to some extent artificial, but is useful for classificatory purposes here.

## 4.1 *Command and control*

While there has been moderate growth in the use of market-based instruments in OECD countries (OECD, 1989, 1995, 1999) most environmental policy remains rooted in ‘command and control’ (CAC) approaches whereby individual emissions sources are given some form of target or standard to achieve. Most of the standards relate to emission levels or to the abatement technologies employed. Climate change policy designed to comply with the Kyoto Protocol is still to be formulated in most OECD countries but the signs are that there will be a mix of CAC measures and market based instruments (MBIs). Hence we first consider CAC-type policies and the extent to which, as currently promulgated, they could account for ancillary benefits (and costs). We then consider MBIs and the ways in which they could account for ancillary benefits.

### 4.1.1 *CAC: Technology-based standards*

Technology-based standards tend to define the way in which most air pollution policy is formulated. A technology-based standard works by defining the ‘best’ technology for a given emission source and then requires that this best technology be adopted. ‘Best’ is defined in terms of the environmental impact. The terminology varies. In the USA the terminology is Best Available Control Technology (BACT) for new emission sources and Best Available Retrofit Technology (BART) for existing emission sources, both in Prevention of Significant Deterioration (PSD) zones, i.e. zones meeting or exceed national air quality standards. For non-attainment areas, i.e. areas where standards have not been met, new sources must meet the lowest achievable emission rate (LAER) and existing sources must meet a reasonably achievable control technology (RACT). Of relevance to the current discussion is that the US technology standards vary in stringency: LAER for new sources would be very stringent and RACT, for example, would be the least stringent. The gradations of stringency reflect the difficulties that emitting sources face in securing compliance – older plant would generally face higher costs than new plant in complying with a standard. In Europe a similar terminology is employed. Best available technology (BAT) is intended to refer to technologies that need to be introduced virtually regardless of cost. The term ‘best available technology not entailing excessive cost’ (BATNEEC) is reserved for technologies which are not the ‘best’ but which can reasonably be introduced given the costs of adopting technologies. The ‘NEEC’ part of the formula serves the same function as the ‘R’ in RACT in the US formulations.



The issue, then, is whether best or ‘reasonably achievable’ technology standards are consistent with consideration of ancillary benefits. In principle, BAT-type standards can be made consistent with ancillary benefit concerns in so far as ancillary benefits relate to associated emissions. The simplest way is to define ‘best’ technology as technology which achieves not only some defined carbon emission target but also other associated emission targets as well. In fact this is how BAT was defined within the UK context of environmental controls in the context of Integrated Pollution Control (IPC)<sup>156</sup>. The essential point, then, is that there is no presumption that BAT for climate-related regulations has to be confined to GHG emission levels. They already are sensitive to multi-media, multi-pollutant impacts.

Once the focus shifts from BAT to ‘practicable’ or ‘reasonable’ technologies, the scope for integrating ancillary benefits into technology based standards tends to decline. This is simply because such standards are less strict than BAT. Less environmentally efficient technologies will therefore be used and we can surmise that the more ancillary benefits are required to be taken into account, the more environmentally stringent the technology will become. Hence BAT is more likely to account for ancillary effects than ‘reasonable’ or ‘practicable’ technology standards.

The potential flexibility of the BAT regime can be illustrated by the current transition in Europe towards the implementation of the 1996 Integrated Pollution Prevention and Control (IPPC) Directive of the European Union. Under IPPC large combustion plants are subject to ‘best available *techniques*’ (which appears to be similar to BATNEEC in intent, but ‘technique’ suggests more flexibility to include, say, changes in management regimes as well as actual abatement technologies). But BAT in this context extends beyond emission standards (‘emission limit values’) to general requirements for waste minimisation and energy conservation. Thus, to some extent, IPPC will already ‘integrate’ multi-media impacts.

What of the other possible ancillary benefits such as technology-forcing and employment? Economic analysis usually suggests that BAT-style standards are not consistent with technology forcing since the emitting source simply has to comply by acquiring an ‘off the shelf’ technology which tends to be prescribed by the regulator. Unless the emitter can itself influence the choice of ‘best’ technologies, it has very little incentive to seek out even better than best technologies. A fairly simple modification of BAT-type regulations could overcome some of this problem by ensuring that BAT is defined by the regulator according to some ‘menu’ of available technologies, but leaving flexibility for emitters to demonstrate that there are even better technologies. The gain to the emitter should be that the even better technologies are cheaper. If they are not, then the emitter reverts to having no incentive to do better than BAT. Some BAT-type regulations, however, are set beyond what is currently achievable, i.e. the regulator deliberately engages in technology-forcing. On balance, the view that BAT is incompatible with technology forcing needs to be qualified by the potential for some incentives to do better than BAT, but the argument is perhaps not a very powerful one.

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IPC was concerned with regulating emissions so that consequent releases to all media – air, land and water – are accounted for. Authorisations for emitting sources are automatically given if the source meets the ‘best environmental option’ (BEO), i.e. the technology meets all the relevant standards. BEO is defined irrespective of the cost of the technology. BPEO (best practicable environmental option) arises when the cost of complying with the BEO is, in some sense, too high. Any emitter then has to justify the BPEO by reference to costs (the ‘EEC’ part of BATNEEC) and to an overall environmental benefit. In effect, some form of cost-benefit analysis is practised, although without the monetisation of benefits. In turn, environmental benefits relate to multiple media and, in principle, global warming impacts and, say, conventional pollutant emissions are included in the environmental benefits. Formal procedures for calculating such multi-media emissions into an Integrated Environmental Index (EIE) exist (HMIP, 1994).

Employment impacts are more complex. Interpretations of concept like BATNEEC vary (Pearce and Brisson, 1993; Pearce 2000). As we have seen, the relevant environmental impacts can be defined to be local, local and regional, or local, regional and global. The concept of ‘cost’ is also capable of differing interpretations. Typically, cost is taken to refer to the capital and operating costs of the abatement technology in question, i.e. the costs are firm-based. Nonetheless, since the technology that is ‘acceptable’ to regulators is determined virtually on a site-by-site basis, the employment impacts of different technology choices is a relevant consideration. Indeed, it is clear that negative employment consequences arising from compliance costs being significant are regarded as an additional component of ‘excessive’ cost. However, in this context the concern is with BAT as something that is likely to reduce employment rather than enhance it. Could the potential for employment creation be regarded as part of a GHG-BAT standard? BAT is, by definition, implemented on a plant by plant basis. Hence the likelihood that a ‘best’ GHG technology at the level of the single plant simultaneously achieves some desirable GHG emissions target, ancillary emissions reductions, and employment benefits seems rather small.

So long as technologies or available, or achievable in the near future, that secure specified gains in GHG emissions and in ancillary pollutants, there appears to be no major obstacle to integrating ancillary benefit concerns into technology-based standards. Virtually by definition, the scope for doing this is reduced the greater is the relaxation of the technology to being one that is ‘practicable’ or ‘reasonable’. The scope for technology forcing under technology-based standards is not perhaps as limited as is often suggested, but there clearly are problems of securing major ancillary gains on this front with such standards. The employment issue is more complex since the chances that prescribed technologies will secure employment gains within the firm do not seem very high.

#### *4.1.2 Ambient and emission standards*

Increasingly, environmental policy is moving towards ambient and emission standards, including standards that are set within the context of agreements between governments and polluters. In principle, standard setting should be informed by an analysis of all the costs and benefits accruing from the standard, whether the costs and benefits are monetised or not. On the ‘fixed coefficients’ approach, any standard set for a non-greenhouse gas pollutant would result in reductions in greenhouse gases, or at least CO<sub>2</sub>. The fixed coefficient approach is not always appropriate however. If a standard is set for sulphur oxide emissions reduction from electricity generation, and that reduction can only be achieved by adopting flue gas desulphurisation equipment (FGD), then one effect is to reduce power station efficiency and increase CO<sub>2</sub> emissions. In this respect, standard setting is more flexible than the technology-based approach (which might have prescribed FGD) because it leaves the polluter to choose how best to meet the standard. But in other respects, standard setting militates against consideration of multi-pollutant effects, and hence ancillary effects. This is because the standards tend to be set pollutant by pollutant. It is important, therefore to set standards on a multi-pollutant basis.

The issue can be illustrated with the ‘CAFÉ’ standards in the USA. The Corporate Average Fuel Efficiency (C.A.F.E) standards date back to 1975 and the US Energy Policy and Conservation Act. Standards were set for passenger vehicles and light trucks in terms of future targets for fuel efficiency expressed in miles-per-gallon. The standards were set in the wake of the early 1970s OPEC oil price hike and hence had no particular environmental motivation (Harrington, 1996). Since the standards appear to have been about twice as effective in improving fuel efficiency as the accompanying fuel price increases due to crude oil price rises (Greene, 1990), the subsequent issue became one of seeing if further efficiency targets had an environmental justification. Di Figlio *et al.* (1990) suggested that a standard of around 34 miles per gallon (mpg) could be justified on the basis of gasoline *cost savings alone*. The actual standard that had been set for 1985 was 27.5 mpg for passenger cars<sup>157</sup>. Thus, a nearly 25% increase in efficiency could be achieved on a *financial no regrets* basis: financial costs would be equalled by financial benefits and no vehicle performance penalty would be incurred. Once environmental benefits in the form of reduced vehicle emissions (NO<sub>x</sub>, VOCs, CO<sub>2</sub> etc) are added to the financial gains, there is the potential for raising the standard further. As it happens, the links between fuel efficiency and emissions is not straightforward in the US case because emission standards also exist for vehicles. But Harrington (1996) shows that the older the vehicle, the closer does fuel efficiency approximate emissions efficiency. The link is virtually non-existent for modern vehicles but very close indeed for vehicles ten years old. But the linkage is itself complex since the age-emissions relationship has more to do with the failure of emissions control systems on older vehicles than with fuel efficiency as such.

The example illustrates several features of standard setting. First, standards are usually set according to a single goal, say an ambient quality standard thought to be consistent with acceptable health or ecosystem effects, or, in this case, some notion of saving on oil import costs. Second, consideration of all the benefits of a standard – in this case the environmental benefits are the ancillary benefits – could easily result in a stricter standard. Third, in practice complex interactions have to be accounted for. In the CAFÉ case, the presence of a different standard, on vehicle emissions, qualifies the presumption that more fuel efficiency results in more emissions reduction. This observation holds for any context where the ancillary emissions are already subject to some form of limit that must be achieved anyway. In such contexts, however, there may be scope for saving on control costs. The central point is that a failure to incorporate ancillary benefits (and costs) into standard setting *adds* to the inefficiency which many economists believe resides in standard setting<sup>158</sup>. If better solutions cannot be chosen for some reason, it is at least incumbent on regulators to set standards on a multi-effect basis.

## 4.2 *Market-based instruments*

The two most widely advocated MBIs for dealing with GHG reduction are (a) carbon/energy taxes and (b) tradable permits/joint implementation.

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<sup>157</sup> By 1988 actual fuel efficiency was about 28 mpg, but it subsequently fell back to some 25 mpg.

<sup>158</sup> i.e. the cost inefficiency of standards relative to market based instruments.

#### 4.2.1 Carbon/energy taxes

Despite significant lobbies against carbon or energy taxes, some eight countries in the EU-15 group of countries already have, or plan to have, such taxes. While taxes on energy that do not discriminate between the carbon content of fuels are clearly inefficient as GHG –reducing taxes, some countries have felt unable to introduce ‘pure’ carbon taxes. Various factors account for this. Practical difficulties are cited in some cases, but in others it is clear that concerns over employment effects in one or other of the fossil fuel supply industries have led to ‘blanket’ taxes being introduced on all fossil fuels. In these contexts, how should ancillary effects be accounted for?

The theoretical answer is comparatively simple and is illustrated in Figure 3. The horizontal axis shows increasing levels of GHG abatement. The marginal global benefit function is assumed to be downward sloping and, for convenience, the marginal ancillary benefit function is assumed to be constant. Thus  $MGB + MAB$  defines the total benefits that result from an individual country’s decision to abate GHGs. The marginal control cost function,  $MCC$ , is assumed to be increasing in GHGs. The ‘optimal’ tax would have been  $t$  in the absence of consideration of ancillary benefits, but becomes  $t^* > t$  once ancillary benefits are accounted for. Effectively, then, the relevant carbon/energy tax is higher once ancillary benefits are incorporated.

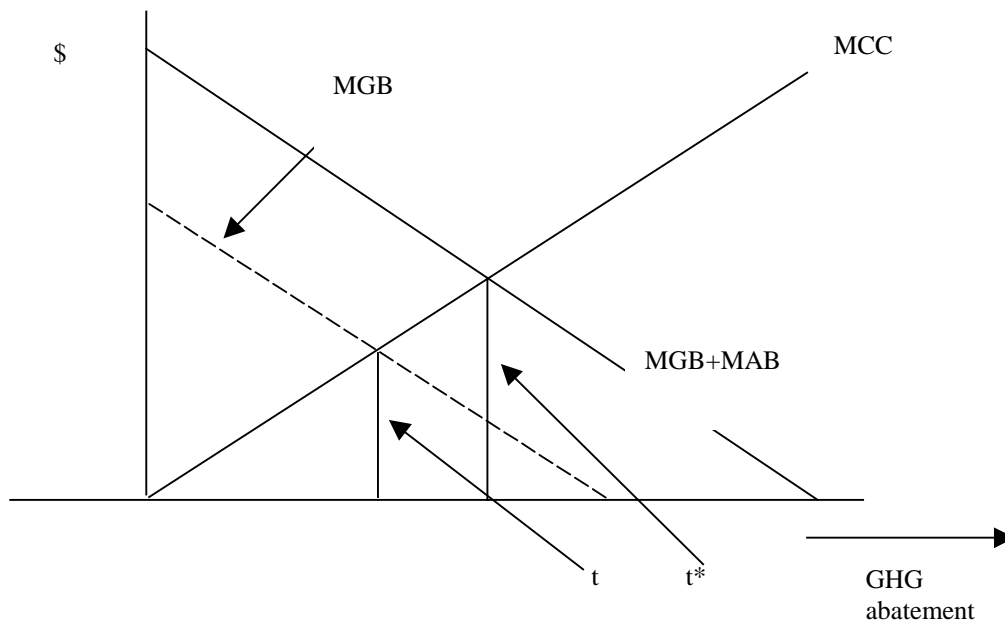
One complication is that the ancillary effects will usually be the subject of separate policies, e.g. to reduce acidification and eutrophication from  $NO_x$ ,  $SO_x$ ,  $NH_4$  and VOCs, to reduce noise nuisance and traffic congestion etc. If the marginal costs of controlling these effects are lower than the marginal ancillary benefits illustrated in Figure 3, then the relevant increment to the carbon/energy tax is given by the avoided control costs rather than the damage costs<sup>159</sup>. Various studies of European policy on the control of conventional air pollutants suggests that marginal benefits exceed marginal control costs for further ranges of controls, despite the considerable advances already achieved by policy (AEA Technology, 1999). These results are, however very sensitive to assumptions about the ‘value of statistical life’ which tends to determine the size of the health benefits accruing from pollution control. If this result was a general one, i.e. all ancillary effects exist at levels below their optimal control, then integrating ancillary benefits into tax design is simpler since it would not be necessary to estimate the ancillary benefit function. All that is required is some knowledge of the abatement cost function for the direct regulation of ancillary effects.

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This point is noted in Ekins(1995, 1996) but Ekins deducts the benefits from GHG control costs. Didactically this obscures the fact that the carbon tax has to increase since the relevant tax still relates to the original abatement cost curve.

Figure 3. Carbon/energy taxes with ancillary benefits



Ekins (1995, 1996) also notes that as abatement technologies for addressing ancillary effects improve, so the ancillary benefit credited to GHG control (i.e. the avoided control costs of ancillary effects) will decline. Whether this effect occurs depends on the extent to which the measures that would be taken to control ancillary effects are themselves ‘technology forcing’. Comparatively little evidence exists on this issue, as was noted in Section 2.4.4 above.

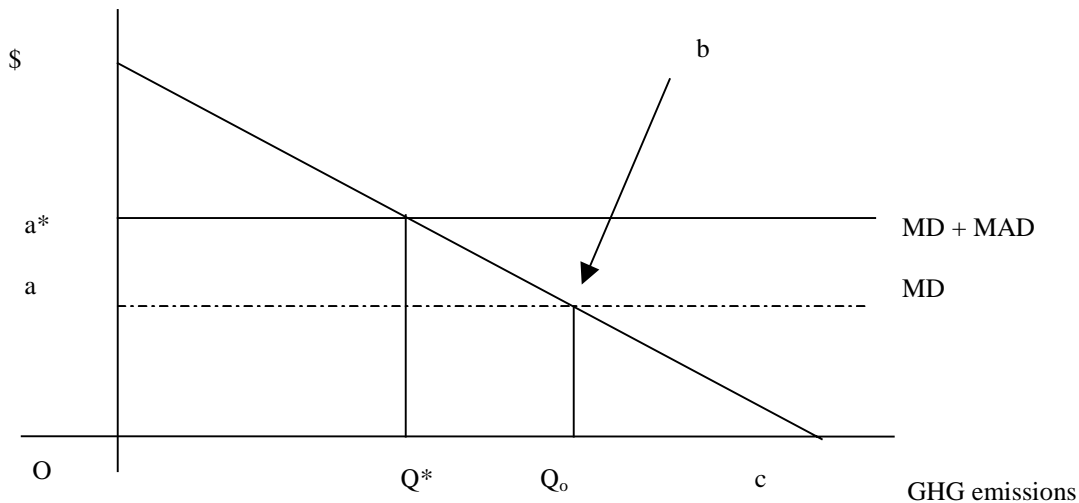
There are several reasons why carbon taxes should not be modified to reflect ancillary effects, however. First, carbon taxes are designed to meet given targets such as those agreed under the Kyoto Protocol. Varying the tax downwards if there are ancillary costs would make the targets more difficult to achieve. Raising the tax would result in ‘overcompliance’ which could be counterproductive if lobbies chose to campaign against the Kyoto targets. Second, ancillary effects are best seen as an added reason in support of carbon taxes (the standard cost-minimisation arguments would perhaps be the main supporting argument). Since such taxes tend to be unpopular, it is beneficial to have a number of rationales for the tax.

#### 4.2.2 Tradable permits/joint implementation

Just as a carbon tax that accounts for ancillary benefit should be higher than a ‘GHG alone’ carbon tax, so the quantitative target set for tradable permits allowances should be stricter once ancillary benefits are allowed for. The essence of the picture is shown in Figure 4. In this case GHG emissions are shown on the horizontal axis, so that control is read from right to left. MCC is the marginal abatement cost curve and  $Q_0$  is the initial allocation of permits. The ruling price of permits is given by the intersection of the vertical supply line through  $Q_0$  and the MAC curve. The emitters are required to purchase from the environmental regulator a  $Q_0$  permits at a cost of  $OabQ_0$ . This sum is a transfer between emitters and the regulator so no real resource cost is involved. In the absence of policy, emitters would emit  $Oc$  tonnes of GHGs. Since they can only emit  $Od$ , the triangle  $Q_0cb$  represents the aggregate abatement costs they incur.

MD is a marginal global damage curve (the mirror image of the marginal benefits curve in Figure 3) and this is shown as constant, just for convenience. Any reduction in emissions therefore saves MD for each tonne of emissions reduced. The optimal level of emission reduction is therefore  $cQ_0$  which we show as being coincident with the result achieved by issuing  $Q_0$  permits. If we also assume ancillary costs are a constant fraction of MD, then the  $MD + MAD$  (marginal ancillary damages) curve becomes the relevant curve for policy. Instead of  $Q_0$  permits being issued for optimality, only  $Q^*$  permits should be issued and the effective permit price should be  $a^*$  not  $a$ . The result is the dual of the carbon tax case since, in the simplest case, optimal carbon tax and optimal permit price will be the same.

Figure 4. Effects of ancillary benefits on tradable permit issues



Of more significance for tradable permits is the effect of ancillary benefits on the distribution of the permits between emitters. Since GHGs are uniformly mixed pollutants the exact location of sources of emissions is immaterial to GHG policy. Ancillary effects are, however, location specific, so that the location of GHG reductions does have an effect on the overall benefit secured. From the emitting country's point of view, the optimal location will be one where the net aggregate benefits of control are maximised. Thus the implications for policy could be considerable since, provided ancillary benefits are significant, it would mean geographically targeting GHG control. One implication could be that GHG control would be best targeted in well populated areas where ancillary benefits per tonne of GHG reduced are likely to be highest. However, the added complication is that abatement costs will vary from one location to another even inside a single country (Bohm, 1997). Hence, per dollar spent, it is the net ancillary benefits that need to be maximised.

The geographical sensitivity of ancillary benefits thus affects trading regimes<sup>160</sup>. Two firms in different locations trading in carbon will affect the damage from ancillary pollutants. If firm A is responsible for higher damages per tonne of, say, SO<sub>x</sub> released, and firm A is the buyer of GHG permits, then the trade will result in higher ancillary costs than would otherwise be the case. The options, broadly, are to ignore these ancillary effects, or to build them into the trading regime as a restriction. An example of the former outcome occurs with the one-to-one trading in SO<sub>x</sub> under the US Clean Air Act. Emitters can trade SO<sub>x</sub> permits even though this may result in damage to a third party who is not part of the trade. This opens the way for legal suit against the parties trading. The alternative is to regulate the trades in emissions in such a way that certain deposition targets must be met (Krupnick *et al.*, 1983; McGartland and Oates, 1985; Tietenberg, 1985). Indeed, such rules are implicitly built into the 1994 Second Sulphur Protocol of the LRTAP governing emissions control in the wider Europe. The rules were designed precisely because cross- boundary sulphur trades might have third country effects<sup>161</sup>.

It is not clear, therefore, how far sensitivities over the ancillary effects of carbon trading will affect the development of these trades. It is not just the localised ancillary effects that matter but the transboundary effects of pollutants such as SO<sub>x</sub> and NO<sub>x</sub>. The fact that no Party to the SSP has so far notified the UNECE of any intention to enter into such trades suggests that, in Europe anyway, such sensitivities are high over sulphur trading and, ergo, may therefore similarly become high for carbon trading. Klaassen (1996) and Bailey *et al* (1996) show that the potential for cost-saving sulphur trades is very modest once deposition constraints are imposed. Potential solutions involve the creation of 'exchange rates' between sources, so that 1 unit of SO<sub>x</sub> in location A could only be traded for X units in location B where X is not equal to 1. But exchange rate systems for sulphur are acknowledged to be administratively very complex and would also probably be controversial. Having an exchange rate system for a carbon trade to reflect not just sulphur but NO<sub>x</sub>, VOCs and even PM would be extremely difficult to imagine.

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<sup>160</sup> It also affects taxes in that optimal taxes will also vary by location once ancillary effects are accounted for. The political sensitivity over the regional impacts of trading regimes appears to be higher (in Europe) than over regional impacts of taxes for reasons that are not entirely clear.

<sup>161</sup> The relevant wording of the SSP changed from explicit reference to requiring that 'environmental improvements for third Parties are not compromised' to requiring that any trades be consistent with the basic obligations on emission reduction and environmental improvements. See Article 2.7 of the Second Sulphur Protocol.

Because trading in carbon has only just commenced on a modest scale in Europe it is too early to say whether the ancillary effects issue will enter into the design and regulation of the trades<sup>162</sup>. Those trades that are taking place – mainly within single companies or between companies within the same sector – appear not to have been influenced by the ancillary effects issue. The most probable reason for this is that the regulations governing ancillary pollutant emissions are strictly binding, i.e. the carbon trades are effectively already constrained by specific regulations on the ancillary pollutants. If so, the potential problems of carbon trades giving rise to adverse ancillary effects will not arise. Indeed, those designing nascent carbon trading regimes have already warned that it is the restrictions on site-specific pollution emissions under Integrated Pollution Control that threaten the potential for carbon trades (Emissions Trading Group, 1999). But, of course, reduced trading could secure ancillary benefits in the sense of ‘beyond compliance’ gains in reduced ancillary pollutants.

#### 4.2.3 *Voluntary and negotiated agreements*

The emergence of ‘hybrid’ policy instruments centring on some form of agreement between polluters and government is one of the most interesting policy developments in recent years. While terminology varies, Börkey and Lévêque (2000) make a useful distinction between *unilateral commitments*, *negotiated agreements*, and *public voluntary agreements*. A unilateral agreement involves a polluter declaring some commitment, e.g. a 5% energy reduction target, a given emissions reduction target, without the involvement of any public authority. A negotiated agreement involves a commitment that is the outcome of a bargain between polluter(s) and government. A public voluntary agreement involves a public commitment, e.g. by a regulatory agency, to which individual firms are invited to participate.

Baeke *et al.* (1999) show that PVAs are most common in the USA, whilst negotiated agreements are most common in Europe. Space forbids a more detailed classification but each category has within it various features which vary according to the individual agreement. Thus, agreements may be *target-based* such that, if targets are met, some alternative regulatory threat is not implemented; *performance based* and primarily aimed at realising unanticipated cost savings and securing ‘green image’; and *co-operative R&D* where government and polluter share the cost of R&D to improve environmental performance. Schemes also vary according to the degree of public involvement, the nature of financial incentives, the sharing of information and so on. For full details see Baeke *et al.* (1999), Mazurek (1998, for the USA), Imura (1998, for Japan) and Börkey and Lévêque (2000, for the European Union) and OECD (2000, for OECD generally).

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World-wide there are around 200 carbon trades dating from the first in 1989. These trades have taken place before the Kyoto Protocol and are not part of the Protocol. Within-firm trades have commenced in the UK in the oil sector.



Another feature of voluntary and negotiated agreements (VNAs) is that they can be linked to other policy instruments. The most obvious way is through a tax as a threat mechanism, but the threat could take any regulatory form, e.g. tighter land use controls, emissions or ambient or technology standards etc. This type of arrangement tends to define negotiated agreements. An example would be the impending Climate Change Levy (CCL) in the UK which is essentially an energy tax, 80% of which can be avoided by implementing an industry-wide package of measures ranging from energy efficiency improvements through to absolute carbon dioxide reduction targets. More subtly, the market instrument may be one of the mechanisms included in the package of industrial measures. Thus there are signs that the CCL regime, when introduced, will be accompanied by within-industry permit trading with any gains from trade being regarded as tradable against the package of environmental obligations.

How, then, would ancillary benefits fit into such mechanisms? Allowing for the fact that the agreements vary enormously in their precise attributes, the general answer must be that such agreements are ideally suited to the inclusion of ancillary effects. First, if the agreement is entirely unilateral, it is open to polluters to declare that their targets include not only greenhouse gas reductions, but ancillary emission reductions as well. Providing the 'fixed coefficients' model pertains – i.e. ancillary emissions are proportional to GHG emissions – industry would in effect be claiming credit for something that would happen anyway, i.e. the ancillary effects are 'free goods'. Nonetheless, it is easy to envisage a gain in 'green image' from such a tactic. In the event that the fixed coefficients model does not pertain, more complex trade-offs would have to be made so that some 'optimal' mix of emissions reductions is secured. Second, if the agreement is a negotiated one, it is open to government or regulator to include ancillary effects in the package of measures required from industry. The advantage of this would be one of ensuring that packages are not negotiated in such a way that ancillary *costs* arise. That is, focusing on ancillary effects in this case amounts to no more than a rational assessment of any package of measures that is advanced by either side. The importance of appraisal mechanisms that ensure this needs emphasis here. Third, many agreements involve internal levies on firms within an industry. The levies may be used for all kinds of purposes but one could well be to advance R&D into further abatement measures, thus stimulating technological development and future primary and ancillary emissions reduction. An alternative would be for government to return some of the revenues from that part of a threatened tax that is actually paid, so as to finance R&D. There are virtually endless possibilities.

Integrating ancillary benefits into VNAs appears unproblematic. The debate surrounding VNAs is in fact a different one, namely whether they are effective or not. First, the emergence of VNAs is partially explained by concerns over the compliance costs of CAC approaches and the difficulties of designing MBIs to replace the traditional regulations. Second, VNAs openly address the issue of asymmetric information, i.e. the view that regulators have only imperfect understanding on the least cost control mechanisms, information that mainly resides with the polluter (Krarup, 1999). Third, and developing the second point, VNAs encourage the sharing of information among firms within a given industry and, ultimately perhaps, between industries. This reduces compliance costs once 'best practice' is identified. This information sharing will occur provided there are few gains from withholding information, which is always a risk. Regulatory costs are similarly reduced. Fourth, there is a potentially significant role for consumer or environmental groups to influence the process, something that can happen only very indirectly with other policy instruments. Interest groups may, for example, help design the package of measures.

Doubts arise from a number of issues. First, as noted, firms may behave strategically with respect to the provision of information setting up barriers to competition rather than reducing compliance costs. This is one of the risks of having the polluters initiate the package and it requires a strong regulator to ensure that these risks are avoided. Second, VNAs are new and the risk of failure is high, where failure can be measured in terms of targets not met or cost reductions not secured. Risks arise from several sources. Firms may well prefer VNAs because it gives them the initiative and they can then lobby the regulator to agree with their stance – a kind of ‘regulatory capture’. Similarly, unless the threats contained in negotiated agreements are activated, or polluters are persuaded they will be activated, there may be a low incentive to comply. And designing packages of measures that contain the right incentives is complex: some VNAs, for example, involve managers achieving environmental management goals which are not linked to actual environmental improvements or to any reward mechanism for achieving the management standard. For these reasons, and many more, quite a few commentators have expressed serious reservations about the effectiveness of VNAs (Nunan, 1999; Bizer and J•lich, 1999). The only real conclusion is that they hold great promise but that some elapse of time is required to see how well they work. As far as climate change ancillary effects are concerned, however, VNAs have more than sufficient flexibility for them to be integrated into the relevant policy package.

### **4.3      *Choosing between technologies***

Choices between technologies (as opposed to abatement technologies now) can be influenced by CAC and MBI measures. We can illustrate with respect to fuel technologies, but the principles are the same for other technologies. Fuel choice is known to be sensitive to price and to direct regulations. How far can regulations on fuel use reflect the concern with ancillary benefits? A good example of the problems is the choice between fuels for goods vehicles. Considerable debate now surrounds the use of diesel fuels versus the choice of compressed natural gas (CNG) and gasoline. Toy *et al.* (2000) suggest that the environmental advantages and disadvantages appear as follows:

<u>Criterion</u>	<u>Diesel</u>	<u>CNG</u>
Fine PM		✓
Ultra-fine PM	?	?
NO <sub>x</sub>		✓
GHGs	✓	
Safety	✓	
Performance	✓	
Cost	✓?	?

Eyre et al (1997) estimate emission factors for vehicles in the UK and find that CNG and diesel are approximately the same in terms of CO<sub>2</sub>. Once upstream fuel cycle activities are included, however, CNG is moderately worse than diesel. If we imagine a carbon reduction policy aimed at fuel choice, the balance may be in favour of diesel. But such a choice would have a significant cost in terms of particulate matter emissions which are known to have serious health impacts. In other words, we have an example of *ancillary costs* rather than ancillary benefits. Targeting technology choice for fuels in terms of a single goal – CO<sub>2</sub> reduction - could therefore be counterproductive. Indeed, Eyre et al (1997) show that, using willingness to pay weights for the damages<sup>163</sup>, CNG is very clearly the preferred fuel in the damage cost ratios gas=1, gasoline = 2.5 and diesel = 3.5. The fundamental conclusion is that climate change policy, just like any other policy, need not be a case of ‘win win’. There will be trade-offs and hence there is a need for risk assessment methodologies to handle those trade-offs. In the Eyre *et al.* analysis the methodology involved is monetary benefit assessment.

The above example can be generalised to other potential contexts where GHG control policy may have the effect of creating ancillary costs. OECD (1999), for example, cites gas-based cogeneration in urban areas where NO<sub>x</sub> increases may result, and the use of nuclear power as a GHG control technology where radiation risks may be involved. Obversely, switches from coal to gas power electricity are likely to yield ancillary benefits. In principle, such comparisons can all be analysed using risk assessment procedures, including cost-benefit analysis.

#### **4.4 Conclusions on ancillary benefits and policy design**

This section has sought to determine whether there are inherent obstacles to the integration of ancillary benefits analysis with the existing modes of environmental policy in OECD countries. As far as conventional ‘command and control’ measures are concerned, technology-based standard setting tends to dominate. This has been strengthened in recent years by the moves towards integrated pollution control where multi-media effects are accounted for. Close inspection of the ways in which ‘BAT’ and ‘BATNEEC’ standards are formulated shows that there should be little difficulty in incorporating ancillary effects into those standards. The reason for this is that the standard setting does often involve multi-pollutant analysis. Thus the development of Integrated Pollution Prevention Control (IPPC) in Europe involves energy conservation standards which already therefore account for multiple pollutants from energy sources. In the UK, the principle of ‘BATNEEC’ already involves multi-pollutant effects.

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<sup>163</sup> Willingness to pay weights are monetary measures of environmental damage expressed in money units per unit weight of the pollutant. These weights are derived from the numerous studies of ‘externality adders’ in Europe and the USA.

As far as carbon/energy taxes are concerned, these have already developed quite rapidly within the EU Member States. Ancillary effects could in principle be incorporated by raising the tax rate (if ancillary benefits exist) or lowering it (for ancillary costs). Since the taxes are likely to be only one instrument among a 'bundle' of instruments for achieving Kyoto targets or targets under the EU burden sharing agreement, there appears to be flexibility in tax policy to achieve this effect. However, there are currently no signs that ancillary benefits analysis has been an integral part of carbon/energy tax design. To a considerable extent this is likely to be because the size of the tax cannot be determined by environmental impacts alone. Perceptions about cost burdens, competitiveness and equity impacts tend to dominate the politics of carbon taxes. As such the tax measures that have been developed, or are being proposed, in OECD countries bear little resemblance to the 'optimal' tax design of economics textbooks. An additional reason for not being concerned with ancillary effects will be the belief that many of these effects, particularly the most studied ones of conventional pollutants, are already well managed by other environmental initiatives.

One of the most promising contexts for integrating ancillary effects into policy packages is through voluntary and negotiated agreements. These VNAs have wide flexibility as to what is included and polluters can readily gain by counting ancillary effects as 'extra' gains from a greenhouse gas target and regulators can easily request that ancillary effects are included. In other words, VNAs have more 'opportunity' for including ancillary effects compared to other policy instruments.

Finally, tradable permit and joint implementation regimes could take account of ancillary benefits and costs (a) through the initial decision on quota issue, and (b) through trading rules that safeguard third party interests. It was suggested however that third party effects, which are perceived as being important in the European context, would be very difficult to integrate into a trading system. This partly explains why the existing sulphur trading regimes do not adopt such safeguards. Again, the belief may be that such safeguards are not required because the relevant pollutants are in any event strictly controlled. Such controls may, however, have the effect of limiting the extent to which carbon trading can take place.

As far as forms of joint implementation are concerned, they too will have third party effects via the interaction with conventional pollutants. In OECD-Europe this may be especially important with respect to any carbon trades between Western and eastern Europe. Trades with developing countries via the Clean Development Mechanism could have deleterious effects on conventional pollution in the investing nations.

## **5. Conclusions and Recommendations**

This paper has had as its main focus the issue of integrating ancillary effects into climate change policy initiatives. From this overview certain conclusions and recommendations can be derived.

1. ‘Demonstrate’ the importance of ancillary effects

Chapter 2 briefly surveyed the estimates of ancillary benefits of climate control policies. There is a very wide range of estimates. Differences would appear to be due to differing methodologies, and different assumptions about ‘policy in the pipeline’ as far as air pollution control is concerned. Nonetheless, even accounting for these differences, the impression remains that ancillary benefits could be comparable in size to the ‘primary’ (global warming) benefits. If so, there is a need to demonstrate these benefits on a more substantial scale. There is little evidence that such concerns have informed existing climate change policies in OECD countries and the ancillary benefits literature has remained largely academic to date (the OECD initiative being the first to broaden the debate).

2. Clearer definition of what ancillary effects are and how they should be presented

Section 2.4 noted that most of the literature on ancillary effects concerns air pollution and a significant part of this literature involves monetary estimates. It could be argued that monetisation both helps and hinders the ‘cause’ of integrating ancillary effects into climate change policies. It should help because it permits direct comparison of the benefits with monetary damage reduction from GHG emissions or with estimated carbon taxes. It may hinder the process if there is hostility towards monetary benefit estimation (see Annex 2 for an overview of the issues). A case can be made for presenting ancillary effects in terms of percentage changes from a baseline, or in terms of probable life years saved, or similar indicators.

The picture is also made more obscure by the inclusion of employment and ‘technology forcing’ effects as ancillary benefits. It is far from clear that climate policies, however formulated, will generate gains in employment, but some models do show this result. More of an issue is the extent to which such effects should be included at all. Section 2.4 notes that it may result in double counting if ‘full welfare analysis’ has been pursued. The literature on technology forcing is very limited and Section 2.4 suggests that what matters in this context is the extent to which climate change policies induce technical change in sectors that are not directly targeted.

3. Play the ‘no regrets’ card

Chapter 3 observed that some climate policy will have ‘no regrets’ features. Definitions of no regrets policies vary from policies where there are actually negative financial costs, through to those that are justified only when ancillary benefits are included. Some economists query whether there really can be negative costs for policy since one would expect the economic system to have taken up those options already. But there is evidence to suggest that information flows and management issues often inhibit the full exploitation of profitable opportunities. Hence no regrets contexts need to be explored thoroughly.

Ancillary benefits can also justify acceleration of climate policies since the ancillary benefits are likely to occur in near time whereas climate change benefits will accrue much later.

#### 4. Use cost benefit approaches where credible

Chapter 3 noted that the cost-benefit paradigm has the potential to account for ancillary effects by simply adding (or subtracting) them from estimates of the primary benefits (i.e. the monetary value of avoided warming damages). As long as the estimates are credible, then, cost-benefit should be used. This is especially important if the primary benefit figures appear not to be supportive of aggressive climate policies since the addition of the ancillary benefit figures could perhaps double the primary benefits. This conclusion is subject to the caveat about the wide range of available estimates of ancillary benefits. Annex 2 explores some of the debates about benefit estimation. It is perhaps especially important to note the debate over the validity of ‘benefits transfer’ since it is this technique that has been used so far in estimating ancillary benefits in terms of reduced air pollution. Explaining opposition to cost benefit would have to be the subject of a separate exercise, but it does need emphasising that this approach is used to very different extents within OECD countries.

#### 5. Use rapid appraisal

Antipathy to the more formal procedures of integrating ancillary effects into policy appraisal may mean that some form of ‘rapid appraisal’ is required. This may be as simple as checklists of likely ancillary effects, or as complex as some form of decision matrix incorporating best available physical estimates of effects. Section 3.4 noted that even these approaches may be difficult in the face of perceptions that what is being integrated are impacts with very different time periods of concern, very different levels of uncertainty and very different solutions (technological versus behavioural change). It is here perhaps that major research effort is required into methodologies for presenting such different impacts together. Ultimately, the ‘reductionist’ approaches such as risk assessment and cost benefit analysis may still be best, but the issue needs more research effort. Arguably, beginning with rapid appraisal can lead on to more formal techniques being used. The risk in using rapid appraisal first is that it becomes the ‘norm’ and there will be resistance to developing it further. As the text noted in several cases, only formal techniques hold out the firm promise of accounting for ancillary benefits due to the discipline involved in identifying costs and benefits.

#### 6. Ensure that standard setting reflects ancillary benefits

As far as conventional environmental policies are concerned, technology-based standard setting tends to dominate. ‘BAT’, ‘BATNEEC’ and integrated pollution control standards are formulated in such a way that it should be possible to incorporate ancillary effects into those standards. Indeed, in some cases they already are built in to the ways the standards are operated in practice.

There may be more problems in the context of emissions and ambient standards because these are often set pollutant-by-pollutant. There are signs that ‘multi-effect’ approaches to standard setting are emerging but the historical record has traditionally not taken this route.

7. Give attention to the role of ancillary effects in carbon/energy tax design

Section 4.2 showed that, in principle, the existence of ancillary effects could be used to redesign carbon/energy taxes to reflect ancillary effects. Carbon taxes could be higher if there are ancillary benefits and lower if there are ancillary costs. However, there are arguments that militate against modifying taxes in this way. First, carbon taxes are designed to meet given targets such as those agreed under the Kyoto Protocol. Varying the tax downwards if there are ancillary costs would make the targets more difficult to achieve. Raising the tax would result in 'overcompliance'. Second, ancillary effects are best seen as an added reason in support of carbon taxes (the standard cost-minimisation arguments would perhaps be the main supporting argument). Since such taxes tend to be unpopular, it is beneficial to have a number of rationales for the tax.

There is currently no evidence to suggest that ancillary benefits analysis has been an integral part of carbon/energy tax design in those countries that have developed such taxes. To a considerable extent this is likely to be because the size of the tax cannot be determined by environmental impacts alone. Perceptions about cost burdens, competitiveness and equity impacts tend to dominate the politics of carbon taxes. As such the tax measures that have been developed, or are being proposed, in OECD countries bear little resemblance to the 'optimal' tax design of economics textbooks. An additional reason for not being concerned with ancillary effects will be the belief that many of these effects, particularly the most studied ones of conventional pollutants, are already well managed by other environmental initiatives.

8. Take advantage of the flexibility of voluntary approaches

Voluntary and negotiated agreements are ideally suited to ancillary benefits analysis because of their flexibility. Where the package of measures is initiated by corporations, ancillary effects can be claimed as 'extra' benefits of the package, even if they are automatic free goods because of a fixed relationship of the emissions with GHG emissions. Where the package originates with, or is developed by, regulators, appraisal techniques should be used to ensure that the package accounts for ancillary effects. Guidance on appraisal techniques is generally issued in the form of handbooks and guidelines by most governments and regulatory agencies.

9. Monitor the ancillary effects of carbon trades

Chapter 4 noted that tradable permits and joint implementation in carbon could have third party effects in terms of ancillary pollutants. It was suggested however that third party effects, which are perceived as being important in the European context, would be very difficult to integrate into a trading system. Additionally, there is a belief that third party safeguards are not required because the relevant pollutants are in any event strictly controlled. Such controls may, however, have the effect of limiting the extent to which carbon trading can take place.

As far as forms of joint implementation are concerned, they too will have third party effects via the interaction with conventional pollutants. In OECD-Europe this may be especially important with respect to any carbon trades between Western and eastern Europe. Trades with developing countries via the Clean Development Mechanism could have deleterious effects on conventional pollution in the investing nations.

The complexities associated with carbon trading arise because what is being jointly produced is a uniformly mixed pollutant (carbon) the location of which does not matter in terms of warming damage, and a local and transboundary pollutant the location of which does matter.

10. Accept that there will be trade-offs

While it is tempting to think that many environmental policies have 'win win' features, the reality is that most do not take this form. There are losers. Section 4.3 looked more closely at one example: the choice between vehicle fuels. It was noted there that diesel might be a preferred fuel if the only concern is CO<sub>2</sub>, but diesel becomes a distinctly inferior fuel once all impacts are accounted for and monetary (willingness to pay) weights are applied.

11. Press for 'joined up' government

The principles of sustainable development require that environment be integrated into all social and economic policy, whatever its nature. The practice is a long way from this goal simply because regulatory agencies and government departments have their own goals and their own bureaucracies. They are not necessarily 'social welfare maximising' entities. Nonetheless, some progress has been made, e.g. by issuing environmental guidance to non-environmental departments, by designating individuals in different departments to be 'responsible' for environmental concerns and so on. This 'greening of government' opens up more possibilities for securing a holistic look at individual policies. But just as there are obstacles across separate departments, so there are obstacles within a single environmental agency or department. Those responsible for climate change policy may be quite separate from those who are responsible for air pollution or traffic control, for example. Communications between divisions involves transactions costs for individuals who tend already to be fully stretched by the demands of policy. Indeed, 'time to reflect' often seems to be missing within operational departments and agencies. The aim of 'joined up' government is to reduce these transaction costs and to get decision-makers to think holistically. The forces inhibiting the 'joining up' are formidable, but the pressure needs to be maintained.



## ANNEX 1. SIMPLE ANALYTICS OF ANCILLARY BENEFITS

Suppose there is a climate change target which, if achieved, would result in (global) benefits of  $B_g$  and costs to the emission-reducing nation of  $C_d$ .  $B_g$  here refers to the *avoided global warming damage* at the global level. Assuming that the emission-reducing country is acting because of its global concerns, it will be more inclined to take the requisite action if  $B_g > C_d$  and less inclined if  $B_g < C_d$ . Any given policy mix of measures to reduce greenhouse gas (GHG) emissions will also secure a given set of ancillary benefits ( $A_d$ ). Including these benefits could alter the decision to abate GHGs since the relevant calculation is now  $[B_g + A_d] >$  or  $< C_d$ .

Some of the ancillary benefits may arise ‘necessarily’ because, whatever the policy measure, the action of reducing GHGs reduces jointly produced pollutants. Thus, a policy targeted at large combustion plant emissions would certainly reduce other pollutants depending on the ‘end of pipe’ technologies already in place. Other ancillary benefits will be instrument-dependent, i.e. they will depend on the *policy design*. Reductions in road traffic in order to control  $CO_2$  could therefore reduce traffic noise, congestion, vehicle emissions, severance etc. But a policy aimed at reducing GHG emissions from road traffic that focused on, say, fuel efficiency, would not reduce congestion or severance and may not reduce noise nuisance either. Hence  $A_d$  above consists partly of a ‘fixed coefficients’ element ( $A_{d,f}$ ) and an element that is a function of policy design ( $A_{d(M)}$ ). The aim of GHG policy could therefore be one of selecting GHG emission reduction targets and policy design so as to maximise the net benefits of control:

$$\text{Max } (B_g + A_{d,f} + A_{d(M)} - C_d).$$

Given the difficulties of any one country setting a unilateral target, the problem could be reformulated as one of choosing a policy design so as to minimise the overall social costs of achieving a given GHG reduction target, i.e.

$$\text{Min } \{C_d - (A_{d,f} + A_{d(M)})\} \text{ for some given GHG target.}$$

Care has to be taken to ensure that the costs of securing ancillary benefits are less with the targeted GHG policy than they would be if there was a policy specifically targeted at securing the ancillary benefits. Thus, if, say, there are benefits arising from reduced acidification due to lower acidification gases, what matters is that the costs of achieving those benefits is less than the costs of achieving them via a targeted anti-acidification policy. This may seem like a redundant caveat since the costs of securing ancillary benefits via a GHG policy are effectively zero. But just as it is theoretically possible to have a ‘negative cost’ policy for GHG reduction so it may also be possible to have a negative cost policy for acidification reduction.

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## **REDUCING GREENHOUSE GASES AND AIR POLLUTION: A MENU OF HARMONIZED OPTIONS**

by State and Territorial air Pollution Program Administrators (STAPPA) and  
Association of Local Air Pollution Control Officials (ALAPCO)<sup>164</sup>

### **1. Executive summary**

The State and Territorial Air Pollution Program Administrators (STAPPA) and Association of Local Air Pollution Control Officials (ALAPCO) developed *Reducing Greenhouse Gases and Air Pollution: A Menu of Harmonized Options* to assess strategies that simultaneously reduce conventional air pollution and greenhouse gases or GHGs (otherwise known as “harmonized strategies”). Utilizing this document, state and local officials can identify and assess harmonized strategies and policies to reduce air pollution and address climate change simultaneously, enhancing both the environmental and economic effectiveness of these efforts.

In recent decades, a concern has emerged that the Earth’s climate is being altered by increased concentrations of GHGs into the atmosphere as a result of anthropogenic (human) activity. The concern is that activities such as the burning of fossil fuels, waste disposal and agricultural and forestry practices may be accelerating the pace of climate change to a rate that natural systems, including humans and other organisms, cannot accommodate. The growing scientific consensus notwithstanding, the United States (U.S.) Environmental Protection Agency (U.S. EPA) does not currently have clear authority to regulate CO<sub>2</sub>, and the U.S. Senate has passed a resolution blocking the ratification of the Kyoto Protocol as currently written. Meanwhile, U.S. GHG emissions rose by over 11 per cent between 1990 and 1997. If the U.S. is to have any chance of meeting its commitment under the Kyoto Protocol (a 7 per cent GHG emission reduction from 1990 levels, on average, between the five year “budget period” 2008 to 2012), states and localities may wish to consider reducing GHG emissions now.

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In continuing to address criteria pollutant nonattainment challenges, state and local officials have the opportunity to capture significant GHG emission reductions. The most effective path for achieving this goal is to ensure that, in obtaining emission reductions needed for criteria pollutant attainment, the applied strategies are ones that also provide GHG reduction benefits, rather than measures that are ineffective or counterproductive from a GHG perspective.

STAPPA and ALAPCO believe it is important to focus on the relationship between GHG mitigation and conventional air pollutant control, because with few exceptions, strategies that mitigate GHGs will also result in reduced emissions of other air pollutants. The most widely recognized harmonized strategies relate to fossil-fueled combustion, the major source of carbon dioxide (CO<sub>2</sub>), as well as a source for particulate matter (PM), nitrogen oxides (NO<sub>x</sub>), sulfur dioxide (SO<sub>2</sub>), carbon monoxide (CO) and air toxics.

The GHGs that are of chief concern include CO<sub>2</sub>, methane, nitrous oxide, hydrofluorocarbons, perfluorocarbons and sulfur hexafluoride. Ozone is also a GHG; therefore, ozone precursors (i.e., NO<sub>x</sub> and non-methane volatile organic compounds or NMVOCs) have an indirect greenhouse effect.<sup>165</sup> This document focuses primarily on CO<sub>2</sub> for two reasons. First, over half of the predicted global warming impacts are expected to result from CO<sub>2</sub>. In 1997, CO<sub>2</sub> emissions constituted approximately 82 per cent of total U.S. GHG emissions.<sup>166</sup> Second, the primary source of this CO<sub>2</sub> is fossil-fuel combustion, an activity that state and local officials address by regulating categories of emission sources.

Each of the source categories that state and local officials address is discussed below, with a focus on effective harmonized strategies for reducing GHGs and other air pollutants simultaneously. A discussion of market-based approaches to implementing these strategies follows these sections. Finally, the implementation of several key harmonized strategies are examined in four case study areas in the U.S., to illustrate potential reductions in GHGs and other air pollutants.

## **1. Sources and associated harmonized strategies**

Air regulation in the U.S. targets primarily large stationary sources, area sources (groups of smaller stationary sources such as residential and commercial buildings), mobile sources (transportation) and other sources, such as municipal solid waste management and agriculture and forestry practices. There are opportunities in each of these source sectors to reduce traditional air pollutants while also achieving significant GHG reductions. In the stationary source sector, the most attractive harmonized strategies involve switching to a lower-carbon or zero-carbon fuel, increasing the efficiency of fuel use, or both. For area sources, from large commercial buildings to small homes, the key harmonized strategies are based on increasing the efficiency of fuel and electricity use. In the mobile source sector, the opportunities lie in increasing the fuel efficiency and reducing the use of motor vehicles. In the municipal solid waste sector, there are significant GHG-reduction opportunities in landfill gas to energy projects and source reduction and recycling. Finally, in the agriculture and forestry sectors, there are considerable GHG-reduction opportunities in manure management and in the sequestration of carbon, the ability of soils and plants to remove carbon from the atmosphere.

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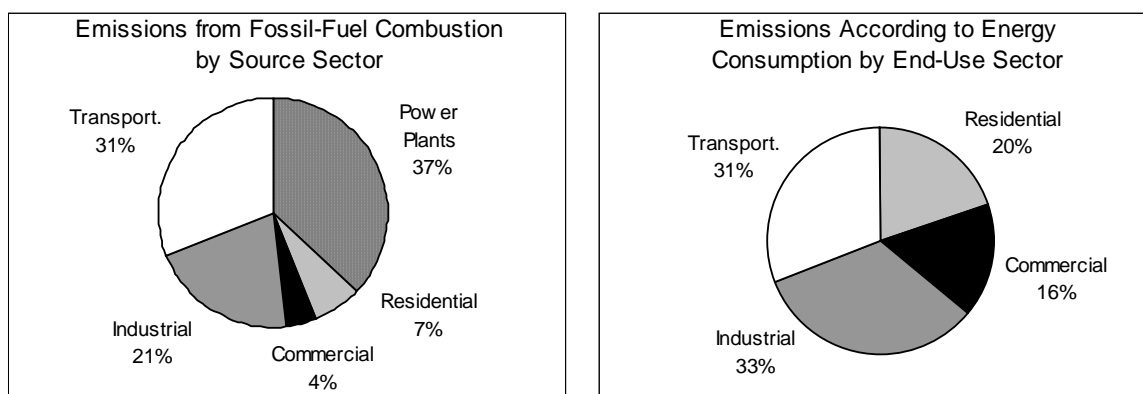
<sup>165</sup> U.S. Environmental Protection Agency (U.S. EPA), *Inventory of U.S. Greenhouse Gas Emissions and Sinks 1990-1997*, Washington, D.C., March 1999.

<sup>166</sup> Ibid.



The generation of electricity is responsible for the largest portion—approximately 37 per cent—of the nation’s CO<sub>2</sub> emissions. The electric industry is also the country’s largest source of SO<sub>2</sub> and one of the largest sources of both NO<sub>x</sub> and airborne mercury. Thus, this industry is an important point of leverage in reducing both conventional air pollution and CO<sub>2</sub>. The transportation industry contributes the second largest share of CO<sub>2</sub> and is projected to be the fastest growing sector, and the other industrial sectors are third. In terms of CO<sub>2</sub> emissions, the primary industrial sectors are the most energy intensive: iron and steel, pulp and paper, chemicals, petroleum refining and cement manufacture. Figure 1 illustrates the portion of total 1997 emissions contributed by each source sector. In the chart at left, power plant CO<sub>2</sub> emissions are shown in a separate category; in the chart at right, emissions are allocated to end-use sectors based on the amount of electricity consumed in each sector.

Figure 1. CO<sub>2</sub> Emissions from Fossil-Fuel Combustion, 1997



Source: U.S. EPA, Inventory of U.S. Greenhouse Gas Emissions and Sinks 1990-1997, 1999.

### 1.1 Large stationary sources

Large furnaces, boilers and combustion turbines constitute the majority of large stationary sources, and in general, these sources are found at power plants and industrial facilities. In both of these sectors, there is enormous potential for reducing GHG and other air pollution emissions, sometimes at a net cost savings.

Air pollutants from large stationary sources can be controlled in familiar ways. Baghouses or electrostatic precipitators can be installed to capture PM less than ten microns in diameter (PM<sub>10</sub>); sulfur emissions can be reduced by switching to lower-sulfur fuels or installing flue gas desulfurization devices (scrubbers) and post-combustion technologies like selective catalytic reduction (SCR) can lower NO<sub>x</sub> emissions. Carbon, however, is a basic component of fossil fuels, not an impurity (like sulfur) or a by-product of combustion (like NO<sub>x</sub>); therefore, removing carbon from flue gases after combustion is energy intensive and extremely expensive. Thus, for the foreseeable future, there are only two practical ways to reduce carbon emissions cost effectively from fossil-fueled combustion: switch to a lower-carbon or zero-carbon fuel or increase plant efficiency so that less fuel is combusted. Fortunately, these operational changes also result in significant reductions of other air pollutants. As a result, the above-mentioned operational changes are effective harmonized emission reduction strategies.

Many of the nation's power plants and industrial facilities are powered by coal, and coal is the most carbon-intensive fuel available. Both oil and natural gas contain less carbon per unit of energy than coal; thus switching a boiler from coal to oil or gas will result in carbon reductions. The magnitude of these reductions will depend on the efficiency of the boiler before and after the alteration. Table 1 illustrates the combined effects of fuel switching and increased efficiency on CO<sub>2</sub> emissions at power plants.<sup>167</sup> Note that emissions in pounds per kilowatthour (lb/kWh) can be reduced by moving across the table (fuel switching), by moving down the table (increasing efficiency), or both.

Table 1. **Approximate CO<sub>2</sub> emissions from fossil fuels**

<b>Plant Efficiency</b>	<b>Heat Rate (Btu/kWh)</b>	<b>Coal (lb/kWh)</b>	<b>Oil (lb/kWh)</b>	<b>Gas (lb/kWh)</b>
20%	17,060	3.53	2.85	2.00
30%	11,373	2.35	1.90	1.33
40%	8,530	1.77	1.42	1.00
50%	6,824	1.41	1.14	0.80
60%	5,687	1.18	0.95	0.67

Source: STAPPA/ALAPCO, 1999.

Chapter II, *Fossil-Fueled Power Generation*, and Chapter V, *Energy-Intensive Industries*, review a number of specific areas in which fuel switching is an attractive option for both emission reductions and cost savings. Perhaps the best example of this opportunity is the gas-fired combined cycle (GFCC) power plant. While coal has historically been the dominant fuel in the electric industry (accounting for 57 per cent of U.S. generation in 1997), falling gas prices and advances in turbine technology have made gas turbines the dominant choice for new capacity in nearly all regions of the U.S.

In addition to replacing the use of coal with gas, the use of excess heat in a heat recovery generator brings the overall efficiency of new GFCC systems to approximately 50 per cent. (Existing coal-fired power plants have efficiencies in the range of 33 per cent.) Together, the fuel switch and efficiency gains offer the following reductions relative to an older coal-fired plant:

- CO<sub>2</sub> – 66 per cent;
- NO<sub>x</sub> – 99 per cent; and
- SO<sub>2</sub> – virtually 100 per cent.

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This table of CO<sub>2</sub> emissions per unit of electrical output is derived from estimates of emissions per unit of heat input developed by the EPA and published in: U.S. EPA, *Inventory of U.S. Greenhouse Gas Emissions and Sinks 1990-1993*, Washington, D.C., 1994. One figure is used in this document for CO<sub>2</sub> emissions from natural gas combustion (117 lb/mmBtu), and a range is given for oil combustion, reflecting different types of oil. The range is from 161 lb/mmBtu for distillate oil to 174 lb/mmBtu for residual oil. For coal, EPA provides 207 lb/mmBtu as a weighted average, reflective of the kind of coal burned in U.S. utility boilers.

Many existing coal-fired plants could be replaced with GFCC capacity at a relatively modest cost. If the entire cost increment of a new GFCC plant were loaded onto CO<sub>2</sub> reductions, these reductions would cost between \$0 and \$39 per ton. Of course, allocating some of the costs of this fuel switch to NO<sub>x</sub> and SO<sub>2</sub> reductions would lower the cost of CO<sub>2</sub> reductions. To put these costs in perspective, estimates of the cost of complying with the Kyoto Protocol range from \$25 to \$150 per ton of CO<sub>2</sub> (see Chapter II, *Fossil-Fueled Power Generation*).

The efficiency of a power plant or industrial boiler can also be increased without simultaneously switching fuels. One of the most attractive options for achieving increased efficiency is the use of excess heat from primary combustion. Excess heat from one process can often be captured and used in another process, removing or reducing the need for a fuel source in the second process. The term “combined heat and power” or CHP is used to describe processes in which electricity and useful heat are produced in the same combustion process (see Chapter II). These CHP strategies can:

- increase overall plant efficiency by 40 to 50 per cent;
- reduce fuel use and all associated emissions considerably; and
- result in emission reductions at a negative cost (or savings) per ton.

Overall, there is tremendous potential for reducing CO<sub>2</sub> emissions by utilizing waste heat in industrial facilities and power plants. The U.S. Department of Energy’s (U.S. DOE’s) recent “Five-Labs Study” estimates that, even without CO<sub>2</sub> reduction requirements in the U.S., power generation at combined heat and power systems is likely to grow to 333,000 gigawatthours per year by the year 2010.<sup>168</sup> If this CHP generation had a CO<sub>2</sub> emission rate 40 per cent below that of conventional coal-fired generation, it would result in CO<sub>2</sub> reductions of 102 million tons per year. This reduction is 4.6 per cent of the decrease (from 1996 levels) necessary to comply with the Kyoto Protocol.

Policies to support fuel switching and increased efficiencies from power plants and other industrial sources include fuel-neutral, output-based emissions standards and comparable emission standards for all facilities.

The move to output-based emission standards, expressed in terms of the amount of pollutant emitted per unit of energy produced, usually pounds of pollution per megawatt-hour (lb/MWh) for CO<sub>2</sub>, NO<sub>x</sub> and possibly SO<sub>2</sub> would incent efficiency enhancements and the use of lower-carbon fuels by making it easier for efficient and cleaner facilities and more difficult for inefficient and more polluting facilities to meet emission limits. These incentives would make it more difficult to operate older, inefficient units and would enhance the value of units with very low emission rates.

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<sup>168</sup> U.S. Department of Energy, *Potential Impacts of Energy-Efficient and Low-Carbon Technologies by 2010 and Beyond*, Interlaboratory Working Group on Energy-Efficient and Low-Carbon Technologies, Energy Efficiency and Renewable Energy Program, Washington, D.C., September 1997.

## 1.2 Area sources

Increasing the efficiency and reducing the use of end-use equipment (demand side management) in the residential and commercial sectors—in contrast to increasing the efficiency of electricity generating units—can vastly reduce GHGs and air pollution emissions. Over one-third of fossil-fuel energy in the U.S. is consumed by the residential and commercial building sectors via lighting, heating, cooling and the operation of appliances. Therefore, the most effective way to reduce air pollution and GHGs from these sectors is to increase end-use efficiency, thereby reducing the amount of fuel consumed directly at the building site and indirectly at the electric generating plant.

The residential and commercial sectors are characterized by a diverse array of energy uses and varying sizes and types of buildings in a wide range of climates. As a result, there is no single method to improve efficiency. Rather, a broad array of technologies are available to reduce GHGs and criteria pollutants through increasing end-use efficiency. These technologies could potentially reduce GHG emissions by approximately 20 per cent, and SO<sub>x</sub> and NO<sub>x</sub> emissions by 20 to 30 per cent in both the residential and commercial building sectors.<sup>169</sup>

The residential sector uses approximately 20 per cent of the fossil fuel consumed in the U.S. Water heating is a main area where energy efficiency can be improved. For instance:

- new low-flow showerheads have a maximum flow rate of half that of older showerheads, and installing one can reduce hot water consumption for bathing by 30 per cent. A new top-quality, low-flow showerhead costs between \$10 and \$20 and will pay for itself within four months;
- leaky faucets and showerheads can be repaired; a leak of one drip per second can cost \$1 per month;
- high-efficiency clothes washers now on the market can reduce hot water use by 60 per cent or more compared with today's average new washer, and by almost 75 per cent compared to an older washer; and
- high efficiency dishwashers can cut hot-water use by about 20 per cent, compared to new machines that are already using about 30 per cent less water than older, existing products.

Also, new lighting technologies and the employment of existing technologies that are intelligently matched to the appropriate lighting needs can achieve significant emission reductions. High-efficiency fluorescent lamps, for example, use less than one-half the energy of incandescent fixtures. Compact fluorescent lamps are another alternative that similarly results in a reduction of energy use in the residential sector. In addition, automatic lighting controls can serve as a supplement or replacement for manual controls.

These strategies have the potential to mitigate GHGs significantly, and as the Five-Labs Study results suggest, most of the strategies will also reduce SO<sub>x</sub> and NO<sub>x</sub>.

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<sup>169</sup>

Ibid.

Similar multiple reductions are also possible within the commercial sector. In the commercial sector, the largest potential for reducing energy use lies in motor drive systems. Motor systems include motor equipment, fans and pumps and transmissions or drivetrains. These systems consume approximately two-thirds of the total electricity in the U.S., and much of this electricity is used very inefficiently. For example, motors are often oversized for their applications, reducing their efficiency. Surveys suggest that about one-fifth of motors above five horsepower are running at or below 40 per cent of rated load. Replacing these oversized motors with smaller, more efficient motors allows the new motors to maintain higher efficiency levels over a wider operating range. In general, optimizing system design, rather than simply choosing individual components, can lead to improvements of 60 per cent using existing technology.<sup>170</sup>

Policies to support increased end-use efficiency include revised building codes and subsidies designed to help overcome market barriers to the adoption of new technologies. Many state and municipal building codes have incorporated more stringent energy requirements in their building codes as a means to reduce energy use. For example, California, Florida, Minnesota and Oregon have developed codes 5 to 30 per cent more stringent than the national Model Energy Code, developed by the Council of American Building Officials.<sup>171</sup> California's Title 24 program is among the nation's most innovative and successful; since 1977, building and appliance efficiency programs administered by the state have saved more than \$11 billion in energy costs.<sup>172</sup>

In addition, most states currently subsidize efficiency upgrades via a surcharge on electricity sales, and in general, these subsidies are being maintained as states move to competitive electric industries.

### **1.3 Mobile sources**

The mobile source sector is responsible for more than a quarter of all GHG emissions in the U.S. High levels of motor vehicle ownership, sprawling land use patterns, limited public transit service, subsidies to the oil industry and low gasoline prices have been major factors in increasing vehicle miles traveled (VMT), and as a result, GHG emissions over the past decade. Since 1990, GHG emissions from transportation have grown by almost 9 per cent. In 1996, the sector was responsible for more than 30 per cent of the CO<sub>2</sub>, more than 40 per cent of NMVOC, 50 per cent of the NO<sub>x</sub> and 80 per cent of the CO emitted in the U.S.

Significant GHG reductions in the transportation sector will require a comprehensive approach that unites technology- and policy-based strategies. In spite of rising GHG emissions from the transportation sector in recent years, there are several reasons to be optimistic. Aggressive efforts are underway at the state and federal levels to reduce urban sprawl and constrain, if not eventually reverse, the steady growth in the use of vehicles. Fuel-efficient and advanced technologies under development by major auto manufacturers and other researchers have the potential to reduce fossil-fuel consumption considerably over time.

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<sup>170</sup> Esource, *Technology Atlas Series* (Boulder, CO, 1997).

<sup>171</sup> Alliance to Save Energy, *Report Card on State Residential Building Codes* (Washington, D.C., 1995).

<sup>172</sup> California Energy Commission (*Title 24: California Energy Efficiency Standards for Residential and Non-Residential Buildings*) 1998. The regulations are available at <http://www.energy.ca.gov/title24/index.html>.

Strategies to reduce transportation-related GHG emissions can address either vehicle emissions per mile driven or the demand for mobility in general. Strategies to reduce emissions per mile driven are generally technology-based. Examples include improvements in fuel efficiency and shifts to new technologies that rely on lower- and zero-carbon fuels. In contrast, strategies to reduce the use of vehicles are generally policy based, such as policies to:

- limit urban sprawl;
- manage traffic; and
- promote use of public transportation.

When the distance traveled per unit of fuel is increased, CO<sub>2</sub> emissions decrease. The U.S. has mandatory fuel-efficiency standards for automobiles, called “Corporate Average Fuel Economy” (CAFE) standards, which require auto manufacturers to maintain a minimum fleet average fuel efficiency for all cars and light trucks sold in a given year. The average fuel economy of the total light-duty fleet has actually declined over the past decade as a result of increasing sales of light-duty trucks and sport utility vehicles, which are held to a lower CAFE standard. Largely as a result of this trend, the overall efficiency of the total light-duty fleet has deteriorated over the past decade.

The U.S. DOE and the Big Three automakers have been involved in the Partnership for a New Generation of Vehicles (PNGV), a cooperative effort to develop a car with a fuel efficiency of 80 miles per gallon. In January 1998, the PNGV selected hybrid-electric vehicles, direct-injection engines, fuel cells and lightweight materials as the most promising technologies to achieve their fuel-efficiency goal.

Another opportunity to lower mobile-sector GHG emissions lies in the use of alternative fuels and advanced technologies, rather than traditional fossil-fueled internal combustion. Of the advanced vehicle technologies, the most promising for near-term commercialization are hybrid electric vehicles (HEVs). HEVs utilize two power sources, and one or both can be used depending on the amount of energy needed. Vehicles combining electric drives with fuel cells or diesel engines hold particular promise.

Progressive vehicle emission requirements at the state level can promote the development of fuel-efficient and advanced vehicle technology by increasing the pressure on automobile manufacturers to develop advanced technology vehicles. California was granted the authority to establish its own vehicle emission requirements by the Clean Air Act. As a result, since 1994, the California Low-Emission Vehicle (LEV) Program has required successively lower average annual emission rates from new vehicles sold in the state and has promoted the introduction of zero-emission vehicles (ZEVs). Other states have aggressively pursued adoption of the California LEV program. The California ZEV sales requirement has spurred tremendous technological advances in electric vehicles and hybrid drive vehicles. The ZEV mandate will require ZEVs to potentially comprise up to 10 per cent of the sales of the major car companies.

Finally, policy-based strategies that reduce the use of motor vehicles are crucial to an overall GHG reduction strategy for the transportation sector. These strategies can focus on:

- land use patterns—encouraging people to live near their workplaces;
- shifting the cost of driving from indirect costs, like annual taxes, to direct costs incurred by actually driving;
- managing traffic to reduce idling time; and
- enhancing public transportation systems.

#### **1.4 *Municipal solid waste***

Municipal solid waste (MSW) management in the U.S. is responsible for a substantial portion of the nation’s anthropogenic emissions of methane, a potent GHG. However, the emissions of criteria air pollutants from the MSW sector are relatively small. As a consequence, while there are many options for reducing GHG emissions from this sector, there are few opportunities for harmonizing these reductions with criteria air pollutant reductions. Opportunities are available, however, for co-control of other pollutants (e.g., hazardous air pollutants from landfills).

The methane emissions from MSW come from landfills, which are the largest single anthropogenic source of methane emissions in the U.S. Municipal solid waste landfills account for over 95 per cent of landfill methane emissions, with industrial landfills accounting for the remainder.

There are two basic approaches for reducing emissions of methane and other gases from landfills.

- Landfill gas can be recovered and either flared or used as an energy source. A system to collect and flare landfill gas will convert virtually all of the methane in landfill gas to CO<sub>2</sub>. Alternatively, the landfill gas may be collected and used for energy recovery. Because methane’s global warming potential is 21 times higher than CO<sub>2</sub>, most of the benefits of those systems are associated with destroying the methane emissions. Simply collecting and flaring landfill gas achieves about 95 per cent of the GHG reductions that are possible by collecting landfill gas and using it for energy recovery. Energy recovery reduces GHG emissions by an additional 5 per cent by displacing higher-carbon fossil-fuel combustion (i.e., oil or coal).
- The quantity of degradable organic waste that is disposed in landfills can be reduced either by limiting the quantity of waste through source reduction or recycling, or by managing the waste in other ways, notably composting. Source reduction and recycling reduces GHG emissions mainly by reducing the use of energy at the manufacturing stage. Composting of organic materials is an aerobic process that avoids the methane emissions associated with anaerobic landfills.

Policy-based strategies in the municipal solid waste sector should be designed to promote recycling, source reduction, composting and other GHG reduction strategies, such as emission trading.

### 1.5 *Agriculture and forestry*

Although the emissions from the agriculture and forestry sectors are relatively low, there are tremendous opportunities in these two sectors to reduce GHGs. Altering farming practices and enhancing carbon sequestration provide two opportunities to reduce GHG concentrations in the atmosphere. Many sequestration opportunities represent “win-win” situations that need only to be identified, publicized and officially encouraged to make significant contributions to both climate change and pollution control efforts. As Chapters VIII, *Agriculture and Forestry* and IX, *Carbon Sequestration* discuss, carbon is constantly moving through the carbon cycle and changes in human activities can increase net storage of carbon in terrestrial systems (thereby delaying or preventing its return to the atmosphere). In many cases it is less expensive to sequester a ton of carbon in biomass than to reduce a ton of carbon emissions. Carbon sequestration can be accomplished in either of two ways:

- increase the rate and amount which carbon is sequestered by living plants; and
- decrease the rate and quantity of decomposition or combustion of existing carbon stocks in soils and forests.

Many industries convert biological waste into usable energy. The same practice can be applied to the agricultural sector. For example, biomass can be converted into gaseous fuel by covering a lagoon filled with animal waste and capturing the gas, primarily methane, as it is produced by the decomposition process. In fact, employing one of these strategies has the potential to reduce methane emissions by 80 per cent on large farms (over 500 dairy cows or 2,000 hogs) in warm climates (see Chapter VIII, *Agriculture and Forestry*). Additionally, using a combination of chemicals and enzymes to break down plant cellulose to sugars that ferment into ethanol can produce liquid fuel. Biomass can also be burned directly to produce electricity, process heat or both. If the energy generated displaces fossil-fuel combustion, emissions of all pollutants, GHGs and conventional pollutants are reduced.

Forests can also be managed to maximize carbon sequestration. One study estimates that between 131 and 200 million metric tons of carbon equivalent (MMTCE) could be offset each year in the U.S. by:

- selecting trees that increase timber growth;
- encouraging longer rotations between harvest cycles;
- ensuring harvesting practices preserve carbon stored in the soil and remaining trees;
- managing forest wastes especially from forest harvests; and
- selecting appropriate uses of prescribed fire.

Policies to reduce emissions of GHGs and conventional air pollutants are only one part of a more complex mix of regulations designed to protect ecosystems. Currently, the areas of environmental regulation that could have an impact on the speed at which carbon is sequestered on U.S. lands include:



- forest management laws;
- water quality programs such as best management practices;
- land use regulation; and
- wetland protection programs.

If emission trading becomes an approved part of the implementation of the Kyoto Protocol, and mitigation credits can be earned by the creation of sequestration projects, the result could be significant financial incentives that would dramatically increase mitigation on the land. Since many sequestration projects result in reductions of both GHGs and other air pollution emissions, the development of these programs is also an important issue for air quality programs.

In order for these trading systems to be successfully adapted to agriculture and forestry programs, several challenges need to be resolved, including:

- development of acceptable methods for measuring the emission reduction values of agriculture and forestry activities; and
- creation of local institutional structures that can work with landowners to install and monitor approved practices, and assemble portfolios of project credits that will be sufficiently large, diverse and credible to attract investors.

Some of these issues will be addressed by the *Intergovernmental Panel on Climate Change Special Report on Forestry and Land Use Change*, due to be released in mid-2000. Decisions based on that report will be very important in establishing the technical framework for implementing any emissions trading or mitigation scheme in both the agriculture and forestry sectors.

## **1.6 Market-based strategies**

Market-based strategies will play a key role in cost-effectively reducing GHG emissions at the local, state, national and international levels. Many state and local agencies are involved with EPA's State and Local Climate Change Program to 1) inventory their GHG emissions; 2) create State Action Plans that identify policy options to reduce those emissions; and 3) implement their state's Action Plan. The policy options recommended so far in these plans are focused on the creation of market incentives to increase energy efficiency, promote alternative fuel and renewable energy use, reduce VMT and internalize the environmental cost of CO<sub>2</sub> emissions.

Market strategies, for the most part, are not sector-specific. Rather, these mechanisms are typically viewed as "cross-cutting" strategies; that is, they can be applied to a variety of sectors, although with varying degrees of effectiveness. There is not a single "one-size-fits-all" market mechanism to reduce GHG that can be applied to every local area and state. Each area has a unique combination of sources contributing to its emissions inventory. As a result, a different mix of market-based strategies will be optimal in different areas. For instance, allowance trading is generally viewed as an effective form of emission trading to reduce GHG emissions from the electricity sector. However, it is less well suited for smaller sources, such as personal vehicles. A better market-based mechanism for smaller, disperse sources might include subsidies for alternative fuels and rebates for the purchase of low emitting vehicles.

Because GHG reductions have not been required in the U.S., little actual experience exists in applying market mechanisms towards the achievement of GHG reduction goals. However, experience with the application of market-based strategies to criteria pollutants provides a useful indication of the issues that are relevant to the application of each mechanism to GHGs.

From a domestic perspective, major source sectors such as electric generators are likely to be targeted with a cap-and-trade mechanism. For example, if the U.S. reduction goal for the electric generating sector were proportional to the reduction obligation under the Kyoto Protocol, then electric generators would have average annual caps for the first budget period (2008 through 2012) set at approximately 450.68 million metric tons carbon equivalent (MMTCE), which is 7 per cent below the sector's 1990 GHG emissions (484.6 MMTCE). If GHG emission levels from the generating sector continued as projected and, by 2010, were to reach a 34 per cent increase over 1990 levels (or approximately 649.36 MMTCE),<sup>173</sup> the emission cap would represent an annual reduction of 198.68 MMTCE or a total of 993.42 MMTCE for the first five-year budget period.

Market incentives have also been used successfully to encourage energy efficiency. The federal government has sponsored energy-efficiency programs for industry and utilities have designed energy-efficiency incentives for potential commercial or industrial energy-efficiency clients.

An excellent example of this concept has been demonstrated by the Indiana Department of Commerce, Office of Energy Policy, which coordinated the design and implementation of a Home Energy Rating System/Energy-Efficient Mortgage (HERS/EEM) program. The HERS/EEM mechanism has two components. The first is a rating system that will classify new and existing homes according to their energy efficiency. This efficiency rating provides estimates of utility costs and may include recommendations for specific energy improvements. The second component allows mortgage lenders to incorporate the lower energy bill expected in a more energy-efficient house when evaluating mortgage applications. The goal of the program is to improve the energy efficiency of Indiana homes and to allow homebuyers to make informed decisions regarding the costs of operating a home.

- By giving regulated sources flexibility in choosing the means of compliance, market mechanisms can allow the target environmental goals to be realized at lower costs, and can encourage innovation as well.

### ***1.7 Harmonized measures – Reducing criteria pollutants and greenhouse gases***

As this document details and this summary has highlighted, there is an important relationship between GHG mitigation and conventional air pollutant control. To evaluate the emission impacts of harmonized strategies, an assessment model has been developed to estimate reductions of criteria pollutants and GHGs in the electricity, commercial and residential, transportation and industrial sectors. It is important to note that the assessment model has been designed to compare the relative magnitudes of emission reductions that can be expected from source sectors in different regions by implementing these strategies.

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<sup>173</sup> U.S. Energy Information Administration, *Annual Energy Outlook 1999*, Washington, D.C., December 1998.

Four areas of the U.S., the state of New Hampshire; Atlanta, Georgia; Louisville, Kentucky and Ventura County, California, serve as case studies for the assessment of selected harmonized strategies. The areas that participated in these case studies are not currently implementing the strategies identified, nor have they committed to implement these strategies. The purpose of these case studies is to begin to evaluate the potential carbon reductions available from comprehensive harmonized strategies.

In most areas, the electric or transportation sector is the largest aggregate emitter of GHGs, with each one typically accounting for 35 per cent to 40 per cent of total emissions. Industrial sources are usually the third largest emitters, followed by the commercial/residential sector. Therefore, harmonized strategies focused on these source sectors. Each area chose its own mix of harmonized strategies, which included:

- switching to natural gas-fired steam generation at an existing coal- or oil-fired unit;
- replacing existing fossil-fueled steam cycle capacity with natural gas-fired combined-cycle capacity;
- replacing fossil-fueled power generation with renewable generation (e.g., wind, solar, hydro and biomass);
- replacing fossil-fueled power generation with primary or distributed fuel cell generation;
- reducing electricity consumption via improved end-use efficiency;
- establishing cogeneration systems at power plants and industrial sources;
- improving transportation fuel efficiency; and
- reducing vehicle use, by increasing such alternatives as carpooling, mass transit and telecommuting.

In aggregate, the results of the model for the four case study areas demonstrate that a range of effective strategies exist that can reduce GHG emissions and also contribute to criteria pollutant reduction goals. The distribution of emission reduction impacts among the four areas is a result of their different emission inventory profiles, their respective nonattainment status for criteria pollutants and the control strategies already adopted or to which the area has already committed.

This analysis indicates that the 7-percent reduction in GHG emissions targeted for the U.S. in the Kyoto Protocol is well within reach of most states and localities. The harmonized control strategies also provide additional criteria pollutant reductions required to meet current and future clean air mandates. Table 2 summarizes the total per cent reductions from baseline emissions that each area would realize with its package of harmonized control strategies.

Table 2. **Percent reduction from baseline emissions in four case study areas**

<b>Area</b>	<b>SO<sub>2</sub></b>	<b>NO<sub>x</sub></b>	<b>PM</b>	<b>VOC</b>	<b>CO</b>	<b>CO<sub>2</sub></b>
New Hampshire	41%	17%	12%	3%	4%	12%
Atlanta, GA area	40%	6%	1%	3%	4%	7%
Louisville, KY area	26%	14%	3%	3%	4%	15%
Ventura County, CA	2%	4%	1%	4%	4%	11%

Source: STAPPA/ALAPCO, 1999.

## 2. **Conclusion**

Many effective opportunities exist at the federal, state and local levels to reduce GHG emissions and, at the same time, achieve substantial criteria pollutant reductions. These strategies are generally technically feasible and cost-effective and can play a substantial role in meeting current and future clean air mandates, including the Kyoto Protocol.

## IV. ANNEX

### EXPERT WORKSHOP ON ASSESSING THE ANCILLARY BENEFITS AND COSTS OF GREENHOUSE GAS MITIGATION STRATEGIES

27-29 March 2000  
Washington, DC

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**EXPERT WORKSHOP ON  
ASSESSING THE ANCILLARY BENEFITS AND COSTS OF  
GREENHOUSE GAS MITIGATION STRATEGIES**

**27-29 March 2000, Washington, DC**

**Resources for the Future Conference Centre**

**AGENDA**

**Monday, March 27, 2000**

**8:00-8:20            Registration**

**8:20-8:30            Welcome and Introduction to Workshop**  
Paul Portney (USA), President RFF

**8:30-8:40            Assessment of ancillary impact of GHG mitigation measures: IPCC  
concerns and policy relevant issues**  
Robert Watson (USA) IPCC Chair

**Framework for Estimating Ancillary Benefits and Costs**

*Chair:* Leena Srivastava (India)

**8:40-8:55            Scope and Purpose of the Workshop**  
Devra Davis (USA) and Alan Krupnick (USA)

**8:55-9:20            *Conceptual Framework for Estimating Ancillary Benefits and Costs.***  
Presenters: Anil Markandya (UK), Alan Krupnick (USA), Dallas Burtraw  
(USA)

9:20-9:55            Discussion: H. Asbjorn Aaheim (Norway)

**9:55-10:2            *How to think about the Baseline?***  
Presenter: Richard D. Morgenstern (USA)

10:20-11:05        Discussion: Joel Scheraga (USA)

**11:05-11:30        Break**

**Chair:** Jean-Charles Hourcade (France)

**11:30-11:55**      *Modelling Ancillary Benefits and Costs*  
Presenters: Jae Edmonds (USA) and Hugh Pitcher (USA)

11:55-12:20      Discussion: Hadi Dowlatabadi (Iran)

**12:20-1:20**      **Lunch**

### **Conceptual and Empirical Issues**

**Chair:** P.R. Shukla (India)

**1:20-1:30**      **Report on TAR Activities**  
Ogunlade Davidson (Sierra Leone)

1:30-1:55      Estimating ancillary impacts, benefits and costs of proposed GHG mitigation policies for public health  
Presenters: Alan Krupnick (USA), Devra Davis (USA) and George Thurston (USA)

1:55-2:20      Discussion: Lester Lave (USA)

**2:10-2:45**      *Estimating ancillary impacts, benefits and costs on ecosystems from proposed GHG mitigation policies.*  
Presenter: Dale Rothman (USA)

**2:45-3:10**      **Discussion: Corjan Brink (Netherlands)**

**3:10-3:30**      **Break**

3:30-3:55      *Methods for Estimating Ancillary Impacts, Benefits and Costs of proposed GHG mitigation policies on Transportation*  
Presenter: Stef Proost (Belgium).

3:55-4:20      Discussion: Philippe Crabbe (Canada)

**4:20-5:00**      **General summary discussion**

**5:00- 5:30**      **Organizational Meeting with Working Group Co-chairs, facilitators and rapporteurs**

**6:00-7:30**      **Reception at World Bank, 1818 H Street N.W. Second Floor Mezzanine.**

**Tuesday March 28, 2000**

*Session Chair:* Bingheng Chen (China)

**8:00-8:10**            **Introduction to Co-control Studies**  
Leland Deck (USA), Abt Associates

**8:10-8:35**            **Ancillary and Co-control Benefits Estimates for Chile**  
Presenter: Luis Cifuentes (Chile)

8:35-9:00            Health and economic values for mortality and morbidity cases associated  
with air pollution in Brazil  
Presenter: Ronaldo Seroa da Motta (Brazil)

**9:00-9:25**            **Mexico**  
Presenter: Julia Martinez (Mexico)

9:25-9:55            Open Discussion

**9:55-10:20**          **Korea**  
Presenter: Seunghun Joh (Korea)

**10:20-10:40**        **Break**

**10:40-11:05**        **China**  
Presenter: Mun Ho, (China)

**11:05-11:30**        **India and Chile: Issues for Developing Country Analysis**  
Presenter: David O'Connor (OECD)

11:30-12:00        Discussion: Leena Srivastava (India)

**12:00-1:00**        **Lunch**

### **Europe Studies**

*Chair:* Gene McGlynn (OECD)

**1:00-1:25**            **Hungary**  
Presenter: Kristen Aunan (Norway)

1:25-1:45            Discussion: Alan Miller (GEF, World Bank)

**1:45-2:10**            **European Union**  
Presenter: Terry Barker (UK)

2:10-2:30            Discussion: Shunsuke Mori (Japan)

- 2:30-2:55**            **Assessing transport impacts in Austria, France and Switzerland**  
Heini Sommer (Switzerland)
- 2:55-3:15            Discussion: Anil Markandya (UK)
- 3:15-3:35**            **Break**
- 3:35-4:00**            **Ancillary Benefits of GHG Mitigation in the U.S.—a case study**  
Dallas Burtraw (USA)
- 4:00-4:20            Discussion: Peter Nagelhout (USA)
- 4:20-4:45**            **Interim report from Canada**  
Presenter: Jay Barclay (Canada)
- 4:45-5:30            Discussion: Maureen Cropper (World Bank)
- 6:30-8:30**            **Working Dinners**

Five Working Groups will meet over dinner to have more in-depth discussions of data gaps and research needs in key areas. The dinner will be served at the Double Tree Hotel, at 1515 Rhode Island Avenue, about a 2-block walk from the RFF. On the next day (Wednesday), the same groups will continue their discussion over the lunch at RFF.

Each group will have a chair or co-chairs and a facilitator (with a laptop). Each group will appoint its own oral rapporteur. The designated rapporteur will report back to the plenary session on Wednesday afternoon.

Working Groups on Data Gaps, Research Priorities and Research Needs:

1. Transportation – Co-Chairs: Tom Roper (Australia), Facilitator: Nasir Khattak (Pakistan)
2. Public Health – Co-Chairs: Luis Cifuentes (Chile) and Jon Samet (USA), Facilitators: George Thurston (USA) and Anne Grambsch (USA)
3. Ecosystems – Co-Chairs: Julia Martinez (Mexico) and Lee Mulkey (USA), Facilitator: Susan Herrod-Julius (USA) and Susan Thorneloe (USA)
4. Land Use – Co-Chairs: Tony Janetos (USA) and Ogunlade Davidson (Sierra Leone), Facilitator: Elizabeth Wilson (USA)
5. Theory and Methods – Co-Chairs: Michael Toman (USA) and P.R. Shukla (India), Facilitator: Leland Deck (USA)

**Wednesday, March 29, 2000**

**8:00-9:00 Meeting of Working Group Chairs: an interim assessment of the previous evenings discussions**

*Chairs:* Bill Rhodes (USA) and Gene McGlynn (OECD)

**Links to Policy-Making**

*Chair:* Joke Waller-Hunter (OECD)

**9:00-9:25 How do Ancillary Benefits affect Policy Instruments and Processes?**  
David Pearce (UK)

**9:25-9:50 Ancillary Benefits in National and Local Policies**  
Ken Colburn (USA)

**9:50-10:15 Ancillary Benefits in the Policy Process**  
Jay Barclay (Canada)

**10:15-11:00 Discussion: Thomas Sterner (Sweden) and Tom Roper (Australia)**

**11:00-11:30 Break**

**11:30-12:00 General discussion**

**12:00-1:30 Working groups lunches meet in the following rooms:**  
Transportation – 5<sup>th</sup> Floor Conference Room  
Public Health – 6<sup>th</sup> Floor Conference Room  
Ecosystems – 4<sup>th</sup> Floor Conference Room  
Land Use – 1<sup>st</sup> Floor Conference Room A  
Theory & Methods – 1<sup>st</sup> Floor Conference Room C  
Outreach – 1<sup>st</sup> Floor Conference Room B

**Others lunch in RFF Courtyard**

*Chair:* Anthony Janetos (USA)

**1:30-2:30 Reports from Working Groups (10 minutes each)**

**2:30-4:00 Breakout Sessions to Plan Follow up Work**

- 1. Future International Co-Control Studies** – Conference Room A  
Coordinators: Leland Deck (USA) and Jane Leggett (USA)

2. **IPCC Lead Authors Meeting, Chapters 8 & 9** – Conference Room B  
Coordinators: Jean-Charles Hourcade (France), P.R. Shukla (India), Terry Barker (UK) and Leena Srivastava (India)
3. **Research Needs and Data Gaps** – Conference Room C  
Coordinators: Bill Rhodes (USA) and John Topping (USA)

**4:00-4:30**

**Plans for Publication, Next Steps and Closing remarks**

Devra Davis (USA)

Gene McGlynn (OECD)

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