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Environmental Costs of Fossil Fuels

A Rapid Assessment Method with Application to Six Cities

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The World Bank

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Contents

Abstract

Among the key external effects of fossil fuel combustion are urban air pollution and changes in global climate. A study of six cities in developing countries and transition economies estimates the magnitude of these effects and examines how various fuels and pollution sources contribute to health damages and other environmental costs. The study develops a simple but robust method for rapid assessment of these damages. By linking the damage to a particular fuel use or pollution source, the method makes possible cost-benefit analysis of pollution abatement measures. The findings show very high levels of environmental damage and reveal large sectoral differences. By far the greatest share of the total damage is that to human health from exposure to ambient particulates, caused mainly by small pollution sources such as vehicles and household stoves. Large industries and power plants account for a smaller proportion of health damage but are the major contributors to carbon dioxide emissions, which have an impact on global climate. The complex relationships between pollution sources and environmental effects highlight the need for a skillful mix of policy instruments built on rigorous analysis. The damage assessment method proposed in this study provides a useful analytical tool that can be easily applied to other urban areas.

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Executive Summary

Worldwide, exposure to the high levels of particulates in urban air causes hundreds of thousands of cases of premature death and respiratory illness. The levels of exposure and the associated health burdens are much higher in low- and middle-income countries than in rich countries. These country-specific problems interact with a growing concern about global climate change, which has no boundaries. Designing policies and measures to combat the adverse environmental effects of fossil fuels is becoming an urgent challenge.

In addressing this challenge, it is essential to take account of the magnitude of the damages attributable to different fuels, sectors, and pollutants. This paper reports on a study of six large cities around the world that suffer from high levels of air pollution: Bangkok, Krakow, Manila, Mumbai, Santiago, and Shanghai. The study adopts a simple but robust method for rapid assessment of environmental damages from various fuel uses. The method is implemented as a simple spreadsheet model that can be easily replicated for other cities and countries and can be further developed or refined as the need arises.

The model covers three categories of damages from fuel combustion: (a) the adverse health effects of exposure to ambient air pollution in urban areas (for example, increased respiratory illness and premature deaths), (b) local nonhealth effects (reduction in visibility; soiling and material damages), and (c) effects on global climate change. The damage assessment method synthesizes the available evidence on the adverse effects associated with high levels of air pollution with evidence of willingness to pay to avoid these adverse outcomes, drawing on an extensive review of a large body of air pollution valuation literature. Damage assessment techniques based on levels of exposure to certain air pollutants are linked to technical information showing how the combustion of fossil fuels in various sectors leads to elevated concentrations of those pollutants in the urban air and to human exposure. Modeling of the linkages makes possible a cost-benefit analysis of pollution abatement options, including a choice of cleaner fuels or fuel switching, across different sources and sectors. The paper explains the method, discusses its underlying assumptions and uncertainties, and presents a summary of recommended techniques and values for assessing damages in future applications.

The purpose of the study is twofold: to develop a rapid assessment model that can be quickly applied to a city on the basis of limited local data while taking account of the key factors affecting the environmental costs of fuels, and to estimate the magnitude of environmental damages and the contributions of various pollution sources to each type of damage for a sample of cities. The cities selected for the cross-country analysis differ in geographic and climate conditions, demographic characteristics, fuel mix and fuel use patterns, the sectoral composition of the economy, and income level. Together, they have a total population of nearly

50 million and represent a span of variables that affect the environmental costs of fuel use. The evidence emerging from this exercise is likely to be representative of the typical situation in many urban areas in developing countries. Although damages alone are not a sufficient foundation for designing specific policies, the magnitude of environmental costs of different natures and the relative significance of various fuel uses in generating adverse impacts convey valuable information for public policy and have important policy implications.

The main qualitative findings of this exercise are as follows:

- The environmental costs of fuel use in large developing country cities can be so high that marginal damage costs are comparable to or, for some fuel uses, may exceed both producer and retail product prices. For the sample of six cities the marginal damages range from 60 percent for unleaded gasoline and 50 percent for fuel oil to more than 200 percent for automotive diesel.
- In highly polluted urban areas local health effects dominate the damage costs from fuel use; global climate change impacts are far less significant. In the six urban areas the social costs of all environmental impacts totaled US$3.8 billion, of which health impacts accounted for 68 percent. Climate change impacts amounted to 21 percent (using a shadow price of US$20 per ton of carbon emissions), and local nonhealth effects contributed 11 percent.
- Vehicles and small stoves and boilers are responsible for more than 70 percent of both the health damages and the total damages from fuel use, while large sources contribute the most to climate change impacts. This implies that policies have to target different sectors and fuel uses and thus may need to be designed differently, depending on whether the primary objective is to mitigate local or global impacts.
- Marginal damage costs per ton of "local" pollutants vary greatly across sources and locations. Because of dispersion and exposure patterns, these costs are much higher for low-level sources such as small stoves and boilers and vehicles than for large industries and power plants.
- The sectoral differentiation in fuel use is at least as significant for the environmental costs of fuel combustion as differences in the type of fossil fuel used. Sectoral differences are driven by variation in combustion and control technologies and by the typical height from which emissions of local pollutants are dispersed. Within a sector, variations in the technology for burning a particular fuel and in fuel quality also yield substantial differences in the environmental costs of the fuel across specific sources.
- Diesel-powered urban vehicles and small stoves or boilers that burn coal, heavy oil, or wood impose the highest social costs per ton of fuel, and these costs are heavily dominated by local health damages. This finding reinforces the importance of promoting a switch by small sources to cleaner fuels, as well as controlling pollution from diesel-powered vehicles.
- Local damage costs greatly exceed global damage costs for all small fuel uses. The gap, however, is largest for urban transport, especially for diesel-powered vehicles. Although transport fuels constitute a significant source of local pollution—45 percent of the local damage from fuel use in six cities—they make a modest contribution of 12 percent to global damage in this sample.
- The large range of environmental damages for different combinations of fuels, sources, and locations limits the efficacy of simple fuel-pricing measures and highlights the need for a skillful mix of policy instruments that can send highly differentiated signals to various users of the same fuels. The main

challenge is to find the right mix for each specific case, taking into account the prevalence of particular fuels and sources in a city and country, existing distortions in fuel markets, and the capacity of governmental institutions. Meeting this challenge requires serious analytical work that integrates damage estimates with an assessment of mitigation options and of the impact of alternative policy measures.

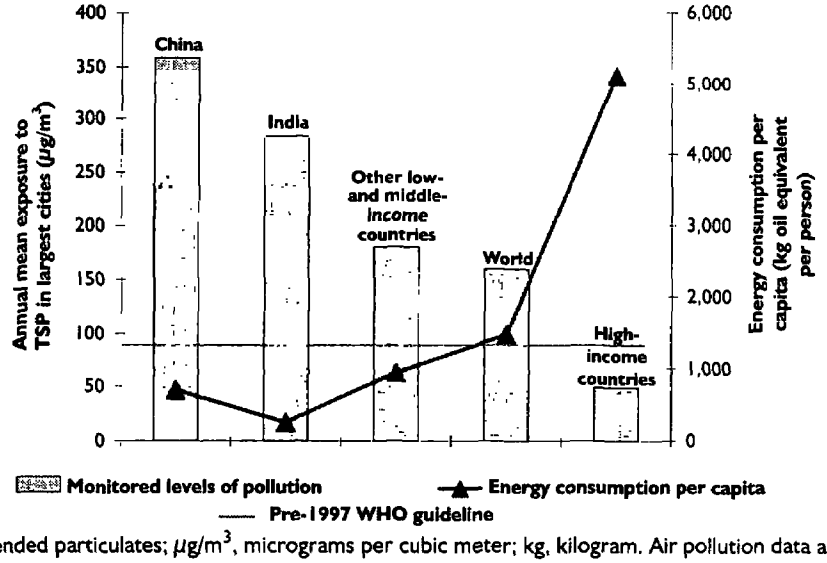
It is important to emphasize that no policy conclusions regarding fuel taxation can be *directly* drawn from this analysis. Fuel pricing and other policy issues will be addressed in a report on a larger study that integrates this damage assessment model with analysis of least-cost abatement strategies and policy measures. The report is under preparation and will be published as a World Bank Technical Paper, *Air Pollution and the Social Costs of Fuels.*

Introduction

The burning of fossil fuels has a number of potential undesirable effects: high levels of urban air pollution; acid rain that damages forests, lakes, and crops; and changes in global climate. Exposure to high levels of particulates in urban air causes hundreds of thousands of premature deaths and millions of cases of respiratory illness worldwide (World Bank 1994; 1997c; WHO 1997). The levels of exposure and the associated health burden in low- and middle-income countries are much higher than in rich countries despite lower energy consumption (see Figure 1). This is because although development fosters energy use, it brings with it policies, technologies, and institutional capacity for combating the adverse environmental effects of fuel consumption on local, national, and regional scales. But the growing consumption of fossil fuels also means more emissions of greenhouse gases, which are believed to increase the likelihood of climate extremes and associated catastrophic events. Thus, country-specific environmental and health problems stemming from fuel use interact with the threat of a change in global climate that has no boundaries.

The design of policies for addressing environmental damages from fuel use at the local, regional, and global levels is a critical

Figure 1 Urban air pollution: A global perspective, 1995



Note: TSP, total suspended particulates; $\mu g/m^3$, micrograms per cubic meter; kg, kilogram. Air pollution data are the most recent available; most of the data are for 1995; energy consumption data are for 1995. The WHO guideline is the pre-1997 maximum value for annual average exposure to TSP recommended by the World Health Organization. (In 1997 WHO waived its guideline values for particulates because of evidence of adverse health effects at lower levels of exposure; no threshold was established.)
Source: World Bank (1998).

challenge for developing countries and transition economies. The experience of industrial countries may not be fully applicable because those countries essentially addressed the impacts of fossil fuels in sequence. First, they focused on local pollution—on the smoky air in cities such as London or Pittsburgh which, at the beginning of the 20th century, were in a situation similar to that of many developing-country cities today. Next, they recognized and sought to mitigate the effects of long-range depositions from power plants burning coal and heavy oil. Now they are turning their attention to global climate change while continuing to tighten controls on local and regional pollution. By contrast, developing countries today face the need to control severe urban air pollution at a time when global impacts can no longer be ignored. The development of efficient policies in this situation requires a new level of analysis that takes account of all these effects.

As a first step in this analysis, it is imperative to know the magnitude of the environmental damages that are attributed to different fuels, sectors, and pollutants. Such knowledge is necessary for performing a cost-benefit test of mitigation options, devising cost-effective abatement strategies, and guiding policy decisions.

This paper develops a simple but robust analytical framework for rapid assessment of environmental damages from various fuel uses. The proposed method for rapid damage assessment is illustrated for six large cities around the world that have high levels of air pollution: Bangkok, Thailand; Krakow, Poland; Manila, the Philippines; Mumbai (formerly Bombay), India; Santiago, Chile; and Shanghai, China. The method is implemented in the simple format of a spreadsheet model that is easily replicated for other cities and countries and can be improved or refined as needed.

The model covers three categories of damage from fuel combustion: the adverse health effects of exposure to ambient air pollution in urban areas (e.g., increased respiratory illness and premature death); local nonhealth effects (reduced visibility, soiling, and material damage); and effects on global climate change. The method synthesizes (a) the available evidence on adverse effects associated with high levels of air pollution and (b) the evidence on willingness to pay to avoid these adverse outcomes, drawn from an extensive review of a large body of literature on valuation of air pollution. Damage assessment techniques based on exposure levels for certain air pollutants are linked to technical information showing how the combustion of fossil fuels by various sectors results in elevated concentrations of those pollutants in the urban air. Modeling of these linkages enables a cost-benefit analysis of pollution abatement options, including a choice of cleaner fuels or fuel switching, across sources and sectors.

This exercise is part of a larger study that integrates the damage assessment model with an analysis of least-cost abatement strategies and policy measures. The report on the entire study is under preparation and will be published as a World Bank Technical Paper, *Air Pollution and the Social Costs of Fuels.*

Although damages alone are not a sufficient foundation for developing specific policies, the magnitude of the environmental costs and the relative significance of various fuel uses in generating adverse impacts convey valuable information. This paper is intended to facilitate discussion of energy and environment issues and to provide a useful tool for analytical work in sector studies and project preparation activities dealing with urban air pollution and fuels.

The paper consists of six chapters and five technical annexes. Chapter 1 gives an overview of the rapid assessment method and discusses the main findings of the study of six cities,

which shows the absolute and marginal environmental damages associated with different fuel uses. Chapters 2 through 5 explain in detail the proposed method for damage assessment. Chapter 2 describes the first two steps, which link fuel use and related emissions of local air pollutants to exposure levels and provide a basis for estimating health impacts and local nonhealth damages. Chapter 3 reviews the evidence of health impacts caused by air pollution and seeks to obtain estimates for these impacts that can be used with sufficient confidence when applied in other situations. Chapter 4 discusses alternative approaches, underlying assumptions, and uncertainties related to valuing health effects and proposes a coherent set of estimates based on willingness to pay to avoid a risk of illness or premature death due to exposure to air pollution. Chapter 5 reviews available estimates of damage costs attributable to local nonhealth impacts and a change in global climate. Chapter 6 summarizes the techniques for rapid damage assessment, analyzes the robustness of the results, and highlights methodological issues that need further research and development.

1 Overview of the Method and the Main Findings

The study seeks to assess the major environmental damages attributable to various sectors, sources, pollutants, and fuels for a number of cities in developing countries and transition economies. This chapter introduces an assessment model and discuss the key findings of the analysis. The following chapters describe the model and the results in more detail.

Rapid Damage Assessment Model

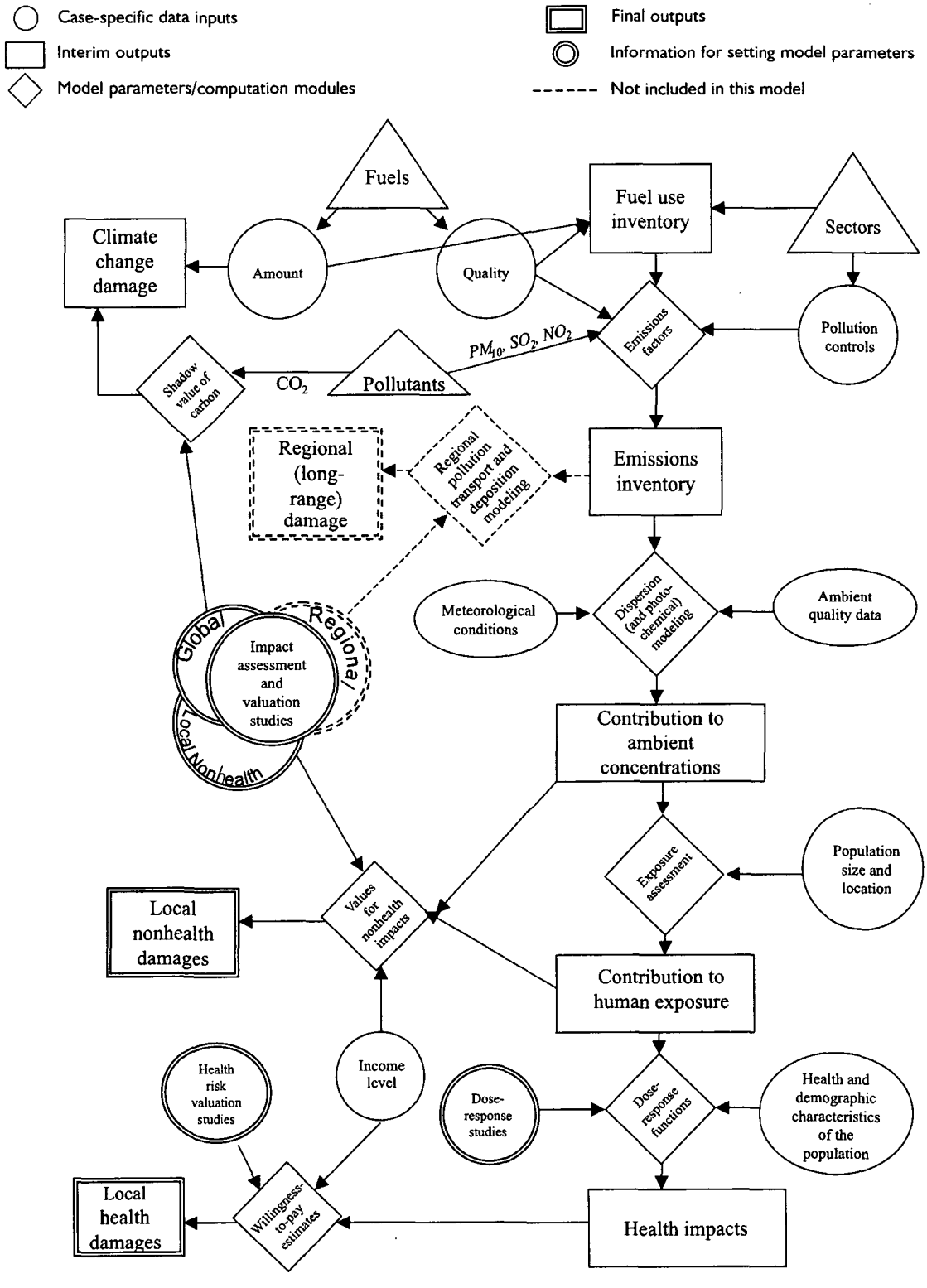
Three categories of damage were considered:

- *The adverse health effects of exposure to air pollution*, mainly in the form of particulates (PM_{10}), in urban areas (e.g., respiratory illness and premature mortality)[1]
- *Local nonhealth effects* such as reduced visibility, soiling, and material damage attributable to elevated ambient levels of particulates, sulfur dioxide (SO_2), and nitrogen oxides (NO_x)
- *Global climate change impacts* associated with emissions of carbon dioxide (CO_2).

These effects are linked to (a) emissions of PM_{10}, SO_2, NO_x, and CO_2 from various economic sectors and (b) fossil fuel use in each sector. The linkages make possible an economic (cost-benefit) analysis of pollution abatement options, strategies, and policies related to fuel quality, combustion technology, alternatives for cleaner energy, and consumption levels. The damage assessment model starts with information on various fuel uses in an urban area and takes this information through the following steps (see also Figure 1.1):

- *Fuel use inventory* of the amount of a particular fuel consumed in each sector of the economy and the quality of the fuel (ash and sulfur content of coal; sulfur content of petroleum products).
- *Emissions inventory* for all fuel uses, based on emissions factors applied to the amount of a particular fuel consumed by each sector (or by a category of pollution sources within a sector), taking account of fuel quality and of any abatement technologies adopted in the sector.
- *Source apportionment* that relates source-specific emissions to effects on ambient conditions and exposure levels. A simple dispersion model with limited data requirements that estimates annual average areawide concentrations of air pollutants was chosen for this analysis (WHO 1989). In addition, simulation of secondary particles from SO_2 and NO_x emissions was introduced (in a primitive way).
- *Assessment of health impacts* using dose-response functions that link variations in the ambient levels of certain pollutants to health effects. The study reviews the extensive literature linking high concentrations of air pollution to adverse health events and seeks to establish a balance of the evidence accumulated to date that can be used with sufficient confidence.
- *Valuation of mortality and morbidity effects* attributable to air pollution. The study uses a coherent set of estimates based on the willingness-to-pay approach that creates a basis for comparison across different effects and countries.

Figure 1.1 Flow chart for the rapid damage assessment model



- *Valuation of nonhealth effects,* including local effects on people's well-being and global climate change impacts.

Four major fossil fuels—coal, fuel oil, automotive diesel, and gasoline—are assessed, as is fuelwood, where its use is significant. The sectoral composition is broken down by five major categories of sources: power plants, large industrial and commercial boilers, small industrial and commercial boilers, household stoves and boilers, and urban vehicles. Diesel is treated separately as a motor fuel only; in other sectors (such as power generation, industrial boilers, and household stoves), it is included in the fuel oil category.[2] In this assessment "fuel oil" is a broad category of liquid petroleum products used in boilers and stoves for various industrial, commercial, and residential needs. The category is dominated by heavy fuel oil but includes some quantities of gas oil (diesel) and light oils. Usually, industries and households in a city use more than one petroleum product; thus, the quality and emissions characteristics of the "aggregate" petroleum fuel (assessed in the model under the fuel oil category) represent a weighted average of all such products consumed within a sector.

Because most fuels are used in several sectors and each sector uses more than one fuel, the model assesses the environmental costs for the following 12 fuel uses, or sector-and-fuel combinations:

Fuel	*Power*	*Large industry*	*Small industry*	*Households*	*Vehicles*
Coal	X	X	X	X	
Fuel oil	X	X	X	X	
Fuelwood			X	X	
Automotive diesel					X
Gasoline					X

The main outputs from applying the model to a particular urban area are:

- Health and nonhealth damage estimates for a 1 microgram per cubic meter ($\mu g/m^3$) change in ambient concentrations of local air pollutants, by pollutant and type of effect (mortality, morbidity, soiling, and so on)
- Damages per ton of emissions, by type of pollutant and sector
- Damages associated with local and global externalities per ton of fuel used, by sector and type of damage.

The approach that apportions health and other social impacts to production activities or fuel use in specific sectors has been applied in a series of studies and has been elaborated in most detail for the social costs of electricity. (See, for instance, Hohmeyer and Ottinger 1991; Ottinger et al. 1991; Pearce, Bann, and Georgiou 1992; World Bank 1994, 1997c, 1997e; Desvousges, Johnson, and Banzhaf 1995; EC 1995; Lee, Krupnick, and Burtraw 1995; Rowe et al. 1995; TER 1995.) This study goes beyond previous work by developing a comprehensive analysis of fuel use in an urban area, together with a rapid assessment model that can be quickly applied to a city on the basis of limited local data while taking account of the key factors affecting the environmental costs of fuels.

Minimum data requirements for this rapid assessment model include:

- Gross domestic product (GDP) per capita, or urban wages
- City population and crude mortality rate
- City size, or population density
- Fuel use by sector
- Fuel quality (sulfur and ash content of coal; sulfur content of liquid fuel)
- The level of pollution abatement at large sources
- Meteorological data or ambient measurements (to validate and calibrate the dispersion model).

The method has been applied to six large cities in different parts of the world: Bangkok, Krakow, Manila, Mumbai, Santiago, and

Shanghai. This exercise provides the basis for similar rapid damage assessments in other cities and countries.

Although all these cities suffer from high levels of air pollution, they differ in geographic and climatic conditions, demographic characteristics, fuel mix and use patterns, sectoral composition, and income levels. Together, they represent a span of different factors affecting the magnitude of the environmental costs of various fuel uses. Thus, the findings emerging from the assessment are likely to be representative of the typical situation in many urban areas of developing countries. It should be noted that in line with the design and objective of this rapid assessment exercise, greater emphasis is given to the evidence from the six-city sample as a whole than to specific details for individual cities.

Table 1.1 summarizes key data for these six cities and shows that there is no simple relationship between ambient air quality and the amounts of fuel used. The main results of

Table 1.1 Key characteristics of the six surveyed cities, 1993

City	GDP per capita (U.S. dollars)	Population (thousands of persons)[a]	Crude mortality (per thousand population)	Ambient air quality ($\mu g/m^3$, annual average)			Fuel use profile (kilograms per capita per year)
				TSP	PM_{10}	SO_2	
Mumbai	300	12,000 (20)	10	207	114	22	Fuelwood: 20 Coal: 80 Fuel oil: 190 Diesel: 20 Gasoline: 20
Shanghai	490	13,500 (2)	7	230	127	74	Coal: 2,150 Fuel oil: 330 Diesel: 30 Gasoline: 60
Manila	950	8,900 (14)	7	177	89	33	Fuel oil: 580 Diesel: 100 Gasoline: 80
Bangkok	2,150	5,900 (4)	7	169	86	13	Fuel oil: 525 Diesel: 100 Gasoline: 140
Krakow	2,260	825 (2)	10	105	58	65	Coal: 4,250 Fuel oil: 4 Diesel: 140 Gasoline: 160
Santiago	3,170	5,200 (18)	6	210	116	38	Fuelwood: 95 Fuel oil: 210 Diesel: 60 Gasoline: 140

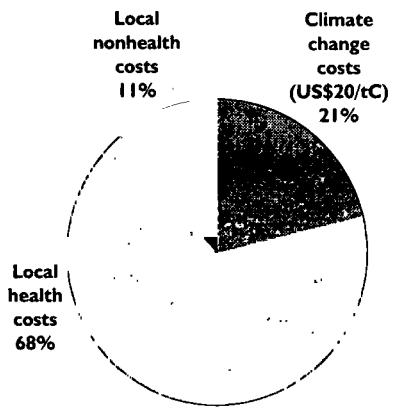
Notes: GDP per capita and crude mortality data are for the entire country. 1992 ambient quality data are used in some cases. When only total suspended particulates (TSP) or PM_{10} data were available, a PM_{10}/TSP conversion ratio of 0.55 was used. Population and fuel use data are for urban agglomerations (e.g., Greater Mumbai; Shanghai District), rather than for city boundaries.
a. Numbers in parentheses are population density, expressed as thousands of people per square kilometer.
Sources: UNCHS (1986, 1996); WHO and UNEP (1992); World Bank (1995b); Annex E in this volume.

the damage assessment are given below. The damage assessment techniques that underlie these findings are extensively discussed and explained in Chapters 2 through 5 and Annexes A through D. Annex E contains complete information on fuel use in the six cities.

The Magnitude and Composition of Environmental Damage

For the six cities, the social costs of all the environmental impacts assessed in the study total US$3.8 billion, with health impacts accounting for the largest portion of the costs for each city. Figure 1.2 shows the shares of the health, "local" nonhealth, and climate change impacts for the six-city sample.

Figure 1.2 Composition of environmental damages from fuel combustion: Averages for the six cities, 1993



Source: Authors' calculations.

In these six urban agglomerations, with a total of 46 million people (1993 data), health impacts due to air pollution from fuel combustion amount to nearly 10,000 premature deaths, 50,000 new cases of chronic bronchitis, and 200 million respiratory illness symptoms per year.[3] These conditions represent a social cost of US$2.6 billion (on the basis of willingness to pay to avoid sickness and premature death), or two-thirds of the total damage. (See Chapters 2 through 4 for an explanation of the methodology for these calculations.) Table 1.2 provides key details for each city.

An important observation on the valuation of local impacts emerging from Table 1.2 is that for some cities the *value* of local damage may be larger than for others even if the physical

Table 1.2 Assessment of the health impacts of fuel use: Six cities

	Mumbai	*Shanghai*	*Manila*	*Bangkok*	*Krakow*	*Santiago*	*All six cities*	*Percent*
Health impacts								
Premature death (number)	2,189	3,979	1,466	822	211	1,054	9,721	
Chronic bronchitis (cases)	7,973	20,709	7,631	4,276	767	6,737	48,094	
Respiratory symptoms	34,808,630	90,407,782	33,314,037	18,335,769	3,350,437	29,412,732	209,629,388	
Restricted activity days	10,937,138	28,406,817	10,467.525	5,761,239	1,052,733	9,241,705	65,867,157	
Social costs (thousands of 1993 U.S. dollars)								
Premature death	73,226	289,930	155,347	197,012	42,477	273,291	1,031,283	39
Chronic bronchitis	32,109	181,617	97,312	123,412	18,626	210,238	663,314	25
Respiratory symptoms	31,630	178,907	95,860	119,405	18,348	207,101	651,251	25
Restricted activity days	11,971	67,712	36,281	45,192	6,944	78,383	246,484	9
Other effects	1,645	11,848	4,986	6,344	1,400	10,773	36,996	1
Total	150,580	730,014	389,787	491,366	87,795	779,787	2,629,329	100
Social costs per urban resident (1993 U.S. dollars per capita)	13	54	44	83	106	149	57	
Social costs as a share of income (percent)	2.8	5.5	3.1	2.6	3.9	4.3		

Note: Based on 1991–93 data; most data are for 1993.
Source: Authors' calculations.

impacts are smaller because the monetary values of damages are linked to the income level in a country or city. This link between income status and the social costs of health impacts helps to explain why, in general, richer countries are willing to adopt stricter and more expensive measures to combat local air pollution. However, the magnitude of the health damages does show that even in the poorest countries, many control measures are very cost effective.

Because of the income adjustment, damage costs expressed as a share of income are better suited for international comparison of the health costs attributable to air pollution. The magnitude of the health costs per average resident in these cities as a percentage of the respective incomes varies from 2.6 percent in Bangkok to 5.5 percent in Shanghai. Since fuel combustion, although significant, is not the only cause of high levels of urban air pollution (see Chapter 2), the overall health costs of poor air quality can be assumed to be even greater than these figures.

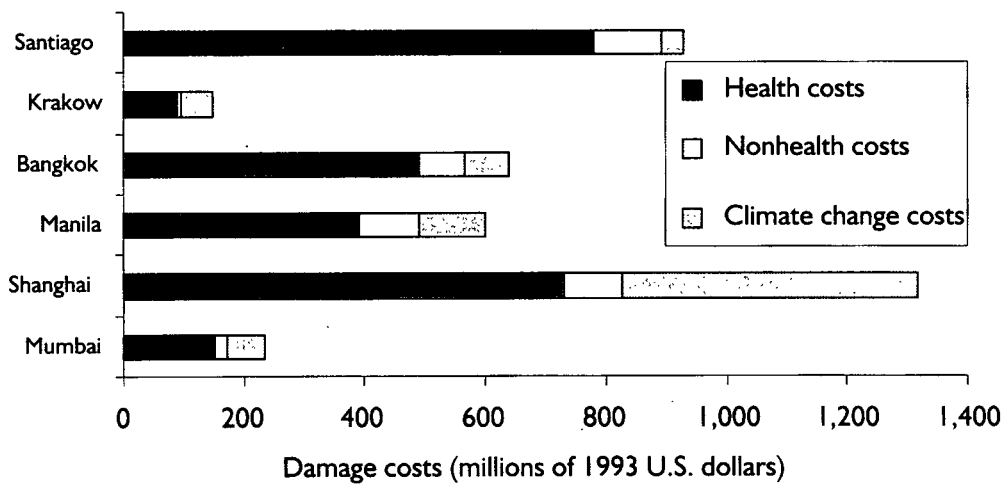
Climate change impacts appear to be a major portion of nonhealth costs (21 percent of the total damage) when a ton of carbon is valued at US$20 (see Chapter 5). Figure 1.3 shows the variations in the damages and the shares of the health, "local" nonhealth, and climate change impacts across cities. Although global damages account for less than half of the health costs imposed by fuel burning in the six urban areas, the situation differs for individual cities. In Krakow and Shanghai, which stand out from the other sample cities because of their high consumption of coal (see Table 1.1), global damages are comparable with the costs of local pollution.

The Roles of Different Sectors, Pollutants, and Fuels

The city-specific fuel mix and the sectoral composition of fuel usage are the key factors that influence the magnitude of the environmental damages and the relative shares of local and global impacts. Figure 1.4 and Table 1.3 illustrate how different sectors contribute to local and global damages and highlight several important points.

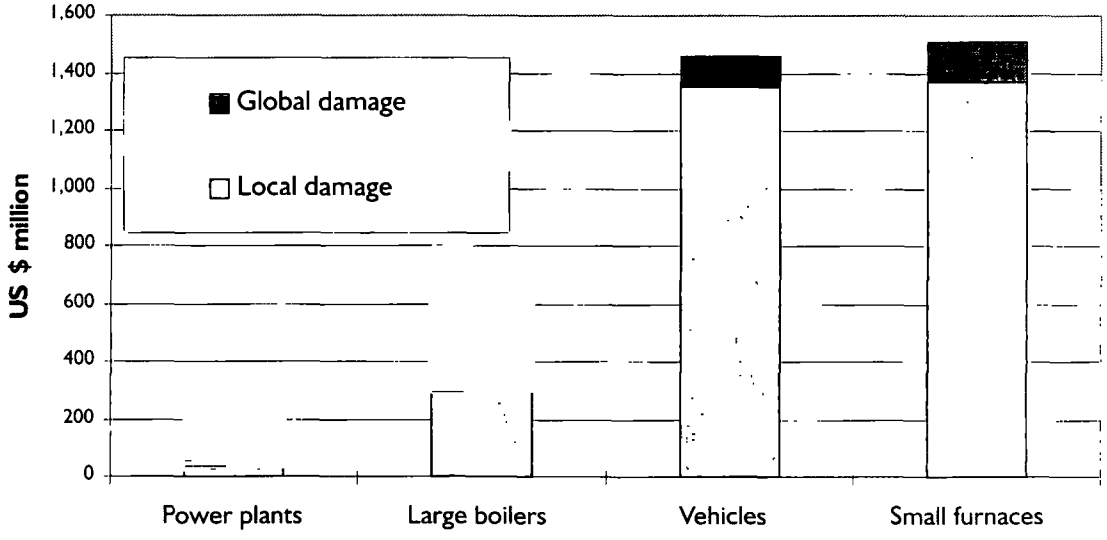
By far the greatest part of the local damages, which are dominated by health impacts, comes from small household, commercial, and

Figure 1.3 Magnitude and composition of environmental damages from fuel use: Six cities, 1993



Source: Authors' calculations.

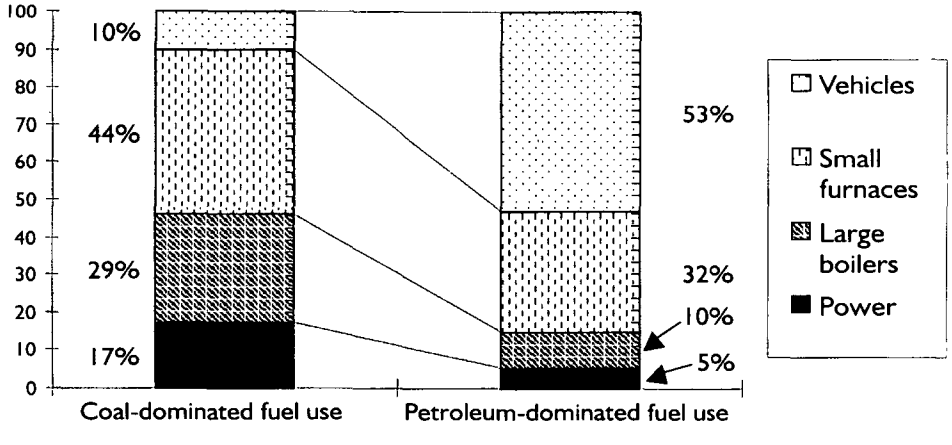
Figure 1.4 Sectoral contribution to local and global damages: Average for the six-city sample, 1993



Note: The category "Large boilers" includes large industrial, commercial, and district heating boilers. "Small furnaces" includes small industrial, commercial, and residential boilers or stoves.
Source: Authors' calculations.

industrial boilers and stoves and from vehicles, rather than from large industries and power plants.[4] Small (low-stack) sources are responsible for much higher local damage costs per ton because the emissions are dispersed over a small area and are very close to the exposed populations, causing a significant increase in both the ambient pollution and the exposure levels. By contrast, large sources, which are the main contributors to CO_2 emissions, have a greater effect on global damages than do small sources. These findings imply that policies have to target different sectors and fuel uses—and thus need to be designed differently—according to whether the primary objective is to mitigate local or global impacts.

Figure 1.5 Sectoral contribution to the environmental costs of fuels in cities with different patterns of fuel use, 1993



Source: Authors' calculations.

Table 1.3 Environmental costs of fuel use, by sector: Six cities (millions of 1993 U.S. dollars)

	Mumbai	*Shanghai*	*Manila*	*Bangkok*	*Krakow*	*Santiago*	*All six cities*
Power plants	21	171	25	n.d.	48	n.d.	264
Health impacts	1	10	4	n.d.	4	n.d.	20
Nonhealth impacts	0	3	2	n.d.	1	n.d.	6
Climate change	19	158	19	n.d.	43	n.d.	239
Large industrial and commercial boilers[a]	28	428	77	70	8	28	640
Health impacts	11	167	28	27	5	16	254
Nonhealth impacts	2	20	10	6	0	3	41
Climate change	15	241	39	37	3	10	344
Small industrial and commercial boilers	99	519	91	65	55	207	1,036
Health impacts	81	434	58	49	48	194	863
Nonhealth impacts	9	43	17	8	4	12	94
Climate change	9	42	17	8	2	2	80
Households	31	102	40	23	16	260	472
Health impacts	19	61	25	16	13	227	362
Nonhealth impacts	2	13	6	2	2	24	50
Climate change	10	28	8	5	1	9	61
Vehicles	55	96	368	480	23	433	1,456
Health impacts	38	58	275	399	17	343	1,131
Nonhealth impacts	9	16	66	57	2	73	224
Climate change	8	21	27	24	4	17	101
All sectors	233	1,317	600	639	150	929	3,868
Health impacts	151	730	390	491	88	780	2,629
Nonhealth impacts	22	96	101	74	9	112	414
Climate change	60	491	110	74	53	38	824

n.d. No data.
a. Includes large district heating boilers.
Source: Authors' calculations.

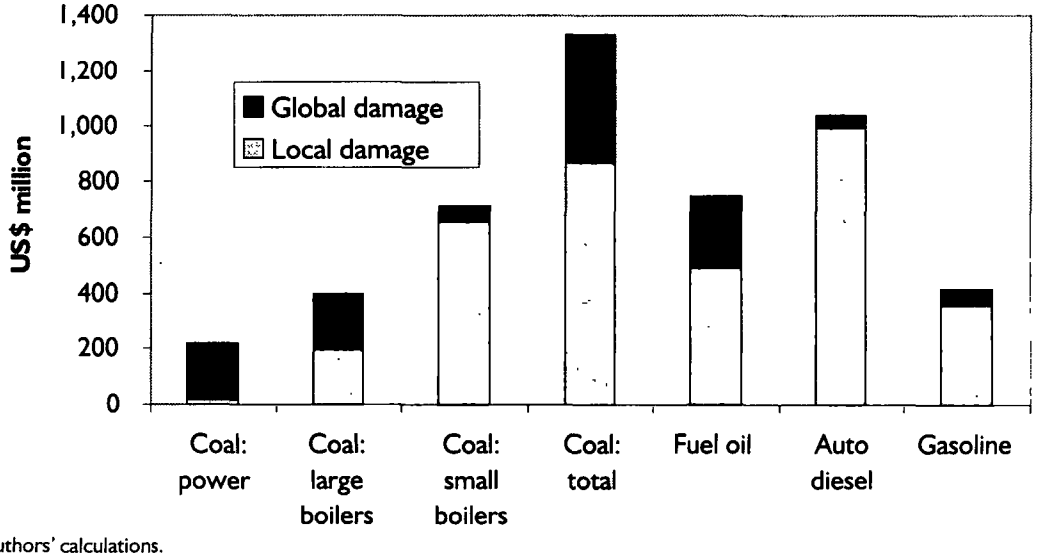
The greatest part—as much as 75 percent—of the total environmental damage in the six-city sample comes from small mobile and nonmobile sources. This conclusion holds true for each city in the sample. What does differ significantly is the relative importance of mobile and nonmobile sources. Figure 1.5 compares the role of different sectors in overall damage from fuel use for two subsets of cities in the sample that are distinguished by their fuel use patterns. One group is dominated by the use of coal (and fuelwood) for various industrial purposes, commercial activities, and residential use. In the other, petroleum products and heavy volumes of traffic dominate. Although vehicle transport accounts for only 10 percent of total environmental costs in the first group, it is responsible for over

50 percent of environmental costs in the second. Many major cities in coal-rich China, India, and Poland are currently in transition from the first pattern to the second. This transition is motivated largely by the high local social costs of coal (and other solid fuels). These costs include, in addition to environmental effects, such factors as the inconvenience and time-consuming nature of solid fuels used for domestic purposes. Another cause for the shift is growth of vehicle ownership as income rises.

Figure 1.6 shows the contribution of different fuels to total, local, and global damage for the sample of six cities and contrasts it with the contribution of different uses of one particular fuel, coal. Quite remarkably, coal, which is not used (or, at least, is not assessed) in half of the cities, accounts for the largest portion of the global damage and the second largest local damage (after automotive diesel) for the sample. In Krakow and Shanghai, where coal is heavily used by households and other small sources, it contributes over 80 percent of the overall damage. Table 1.4, which gives the environmental profiles of each fuel, shows that fuelwood also imposes significant local environmental costs despite its very modest use in the studied sample of cities. Since the most substantial portion of these damages comes from small furnaces, the assessment indicates that an environmental priority should be to promote a switch by small sources from wood, coal, and heavy oil to cleaner fuels, as well as to control pollution from diesel-powered vehicles.

An important observation from Figures 1.4 and 1.6 is the striking disparity between local and global damages from urban transport, especially diesel-powered vehicles. Transport fuels, while a significant source of local pollution, make only a modest contribution of 12 percent to global damage in this sample of cities. Given the small overlap between cost-effective programs for controlling local and global pollution in the transport sector (see, for example, Eskeland and Xie 1998), the comparison of damages suggests that environmental policies for urban transport in developing countries should be driven by local pollution problems. As Figure 1.6 illustrates, coal and fuel oil are the two fuels that may

Figure 1.6 Fuel composition of local and global damages: Average for the six-city sample, 1993



Source: Authors' calculations.

Table 1.4 Environmental costs of fuel use, by fuel type: Six cities (millions of 1993 U.S. dollars)

	Mumbai	*Shanghai*	*Manila*	*Bangkok*	*Krakow*	*Santiago*	*All six cities*
Coal	89	1,115	0	0	126	0	1,331
Health costs	70	644	0	0	71	0	784
Nonhealth costs	6	73	0	0	7	0	85
Climate change costs	14	399	0	0	48	0	461
Fuel oil	71	105	232	159	1	182	750
Health costs	27	28	115	92	0	143	406
Nonhealth costs	7	7	35	16	0	21	87
Climate change costs	37	70	83	50	0	18	258
Automotive diesel	41	63	300	348	18	272	1,042
Health costs	31	46	238	302	14	231	863
Nonhealth costs	6	10	47	36	2	36	136
Climate change costs	4	8	15	10	2	5	43
Gasoline	14	33	68	132	6	161	414
Health costs	7	12	37	97	3	112	268
Nonhealth costs	3	7	19	21	1	37	88
Climate change costs	4	14	12	14	2	12	58
Fuelwood	18	0	0	0	0	313	332
Health costs	16	0	0	0	0	293	309
Nonhealth costs	1	0	0	0	0	18	19
Climate change costs	2	0	0	0	0	3	4
Total damage	233	1,317	600	639	150	929	3,868

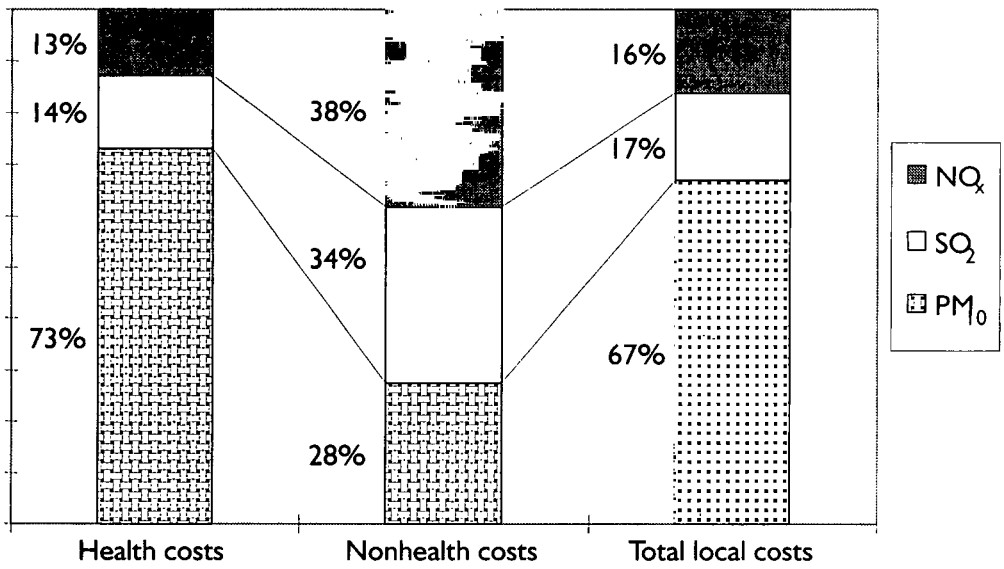
Note: Numbers may not sum to totals because of rounding.
Source: Authors' calculations.

have a considerable potential for overlap between measures designed to deal with local and global issues. Still, the sectoral differences shown in Figure 1.4, together with the various possibilities for controlling local pollution by improving combustion technology or the quality of fossil fuels burned, indicate that the overlap for these fuels may be also limited.

There are tradeoffs not only between local and global issues but also between measures for controlling different local pollutants. It is therefore important to know how the environmental costs of fuels are allocated among these pollutants. Figure 1.7 shows that particulate emissions (PM_{10}) bear a far larger responsibility for health and total local damages than emissions of SO_2 and NO_x, which contribute more to nonhealth effects. It should be emphasized that the estimates for almost all the health damage and a portion of the nonhealth damage from SO_2 and NO_x emissions are based on their contribution to the formation of secondary particulates. (See the discussion on dispersion modeling in Chapters 2 and 3.)

Table 1.5, which shows marginal damages per ton of emissions for PM_{10}, SO_2, and NO_x,

Figure 1.7 Contribution of emissions of various pollutants to local damages from fuel burning in the six cities, 1993 (percent)



Source: Authors' calculations.

demonstrates the large variation in these damages across sources and locations. Small (low-stack) sources of air pollution—vehicles, household stoves, and small industries and businesses—have a far greater impact on ambient quality and the associated environmental costs per ton of emissions. Differences across locations depend on such factors as the size of the city, population density, dispersion conditions, and income level. Given the high sensitivity of marginal costs of pollutants to these factors, it does not seem possible to assign a meaningful uniform externality value to the emissions of "local" pollutants. Instead, these values, even as rough proxies, should be derived for each location and source (or group of sources), taking account of the most important variables. A similar conclusion emerges from a review of several analyses assessing the social costs of electricity (Krupnick and Burtaw 1996).

Environmental Costs and Fuel Prices

Figures 1.8 and 1.9 compare the magnitude of the local and global environmental costs of selected fuels in the study cities with the market prices of the same (or similar) fuels.[5] Tables 1.6 and 1.7 give this information for individual cities and fuel uses. As Figure 1.8 and Table 1.6 show, for average fuel consumption in the six cities, the marginal local environmental costs of fuels are comparable to international market (wholesale) prices—from 60 percent of the market price for gasoline and 50 percent for fuel oil to more than 200 percent for diesel. The marginal costs of local (health plus nonhealth) effects are greater than the marginal costs of global impacts for all fuels. The difference is especially significant for motor fuels and widens to nearly 20 times for diesel. The impact of diesel on the health and well-being of the urban population is greater than the global impact by a large margin for each city. In some cities, however, the global impacts of coal and oil use outweigh the local impacts.

Automotive diesel has the highest marginal cost of any of the fuels in their typical uses for each city. It is important to stress, however,

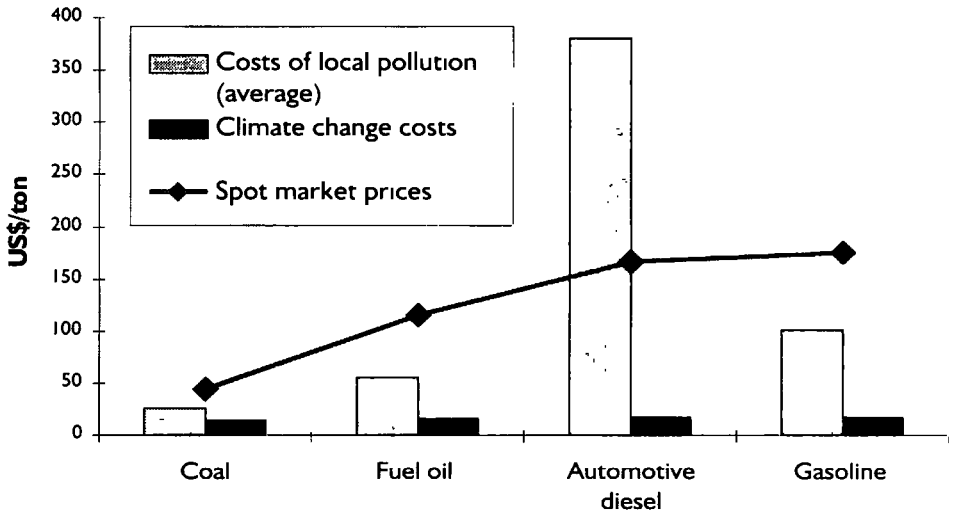
Table 1.5 Marginal damage costs, by pollutant and source: Six cities (U.S. dollars per ton)

City and pollutant	*High stack (power plant)*	*Medium stack (large industry)*	*Low stack or low-level (small boilers and vehicles)*	*Average across fuel uses*
Mumbai				
PM_{10}	234	1,077	7,963	5,137
SO_2	51	236	1,747	549
NO_x	20	93	688	450
Shanghai				
PM_{10}	161	502	5,828	1,562
SO_2	36	112	1,295	253
NO_x	11	33	385	106
Manila				
PM_{10}	345	1,828	17,942	11.710
SO_2	61	324	3,183	612
NO_x	24	129	1,265	997
Bangkok			2	
PM_{10}	828	2,357	28,722	21,010
SO_2	147	417	5,087	1,443
NO_x	57	162	1,971	1,832
Krakow				
PM_{10}	97	682	13,255	1,653
SO_2	18	130	2,522	188
NO_x	4	29	560	107
Santiago				
PM_{10}	692	4,783	88,551	74,906
SO_2	132	911	16,864	6,647
NO_x	35	240	4,445	4,021
All six cities				
CO_2	5	5	5	5

Note: PM_{10}, inhalable particles; SO_2, sulfur dioxide; NO_x, nitrogen oxides; CO_2, carbon dioxide.
Source: Authors' calculations.

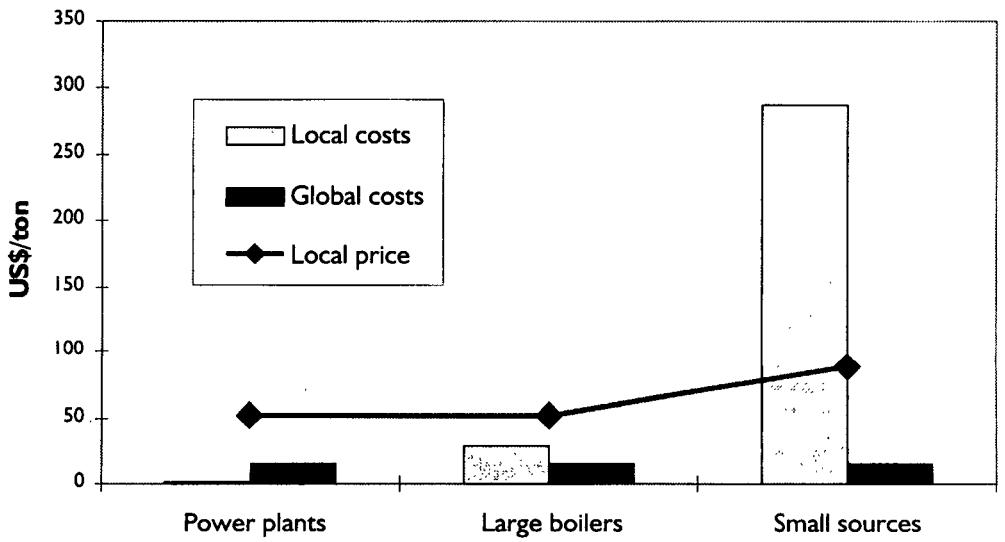
that these high social costs are associated only with diesel used in urban transport and that the rather poor characteristics of diesel fuels and vehicles in the studied sample lead to high emissions of particulates (see Chapter 2 and Annex A). These high costs do not apply to diesel used in power generators, railway transport, and agriculture. Furthermore, not all diesel fuels and vehicles are the same. Even in urban transport, diesel will not impose nearly as high environmental costs as in the sample if "clean" diesel with low sulfur content and other improved characteristics is used in well-controlled new vehicles. However, the reality

Figure 1.8 Marginal environmental costs of fuels: Average usage in the six cities, 1993



Source: Authors' calculations.

Figure 1.9 Marginal environmental costs of coal, by sector: Krakow, 1993



Source: Authors' calculations.

in many developing-country cities is that rapid motorization and dieselization bring with them substantial environmental and health damages.

The environmental costs of coal and oil use in various sectors have to be considered in light of their sectoral differences, as is illustrated for coal by Figure 1.9 and Table 1.7. Coal burned in small, low-stack boilers and in stoves with no controls causes local damage with marginal social costs as high as or higher than those of automotive diesel. (Compare Tables 1.6 and 1.7 for the environmental costs of diesel and of coal burned by small sources in the same cities.) These costs substantially exceed retail coal prices. The marginal local damage from burning coal in large combustion plants in the

Table 1.6 Environmental costs per ton of fuel: Six cities, 1993 (U.S. dollars per ton)

	Coal	*Fuel oil*	*Automotive diesel*	*Gasoline*[a]	*Fuelwood*
Local costs, 1993					
Bangkok	n.d.	35	563	143	0
Krakow	22	192	138	25	0
Manila	n.d.	29	318	78	0
Mumbai	76	15	151	41	57
Santiago	n.d.	150	847	201	628
Shanghai	25	8	123	22	0
Weighted average	26	56	380	101	416
Climate change costs (at US$20 per ton carbon)	14	16	17	17	6
Spot market price, 1993	44	115	166	175	
Percentage of producer price					
Local costs	59	49	229	58	
Climate change costs	32	14	10	10	

n.d. No data.
a. Damage costs for leaded gasoline do not include the effects of lead and ozone.
Source: Authors' calculations.

Table 1.7 Environmental costs per ton of coal, by sector: Selected cities, 1993 (U.S. dollars per ton)

City	*Mumbai*	*Shanghai*	*Krakow*
Local costs, 1993:	76	25	22
Power plants	2	1	2
Large industrial & commercial boilers	22	11	29
Small sources (households and small industrial & commercial boilers)	191	141	296
Climate change costs (at US$20/tC)	*14*	*14*	*14*
Local price, 1993			
for power/industry	6 to 24	37	52
for households/small boilers		39	88

Source: Authors' calculations.

power and industrial sectors is much lower because of a combination of high-stack dispersion and the use of controls to reduce dust emissions. Since large combustion plants account for most of the coal used in these cities, the *average* local environmental cost per ton of coal is relatively small in comparison with the damage per ton for small sources.

Similar sectoral variations are observed for fuel oil. The environmental costs of fuel oil

used by small sources are as large as those of automotive diesel (or larger, for heavy oil with higher sulfur content). The average environmental cost per ton of fuel oil across all sources is far lower.

An important conclusion is that the sectoral differentiation in fuel use is at least as significant for the environmental costs of fuel combustion as are the differences in the types of fossil fuel used. The greatest disparities between marginal environmental costs and market prices of fuels, as well between local and global damages, occur for such fuel uses as urban transport, small industrial and commercial boilers, and household heating and cooking.

Sectoral differences are driven by differences in combustion and control technologies and in the typical height from which emissions of local pollutants are dispersed. Within a sector, variations in the technology for burning a particular fuel and in fuel quality specifications also lead to substantial differences in the environmental costs of the fuel across specific sources.

In addition to the effect of sectoral and technological differences, the environmental costs of fossil fuels are heavily dependent on the locational context—the size, income level, and health status of the exposed population and the meteorological conditions of the area that affect dispersion patterns. These factors further explain substantial variations in the marginal costs of fuels, even for similar fuel-and-sector combinations in different cities (Tables 1.6 and 1.7).

The wide range of environmental damages for different combinations of fuels, sources, and locations that emerged from this exercise shows how difficult it is to devise practicable policy measures to internalize these damages. Obviously, the high levels of environmental costs attributed per ton of fuels do not mean that fuel prices should be adjusted to incorporate these same levels of costs. Simply adding damage-based environmental taxes to fuel prices is neither feasible nor cost-effective, given the high degree of differentiation in the environmental costs of a particular fuel across sources, technologies, and locations.

It is important to emphasize that no policy conclusions regarding fuel taxation can be drawn *directly* from this analysis. First, fuel taxes are based on a variety of considerations, the most important of which are fiscal objectives and social priorities. In countries where addressing the environmental costs of fuel use is considered a social priority, including members of the Organisation for Economic Co-operation and Development (OECD), environmental concerns can influence—but can never fully determine—the design and levels of taxes (see OECD 1996). Second, environmental damages alone—without information on possible mitigation measures and consumer responses to price signals—do not provide sufficient guidance for setting taxes. Even when fuel taxes are indeed motivated, to some degree, by environmental damages, their levels are not set on the basis of environmental costs per ton of fuel. The costs of mitigating these damages are a better foundation for determining the levels of taxes or of other economic incentives such as pollution or user charges.

Particular care should be taken in interpreting the results in this analysis that were obtained for certain uses of multisectoral fuels. An interesting example is that of diesel (gas oil), discussed above. Although the environmental damages from diesel vehicles estimated in this study are much higher than those from gasoline vehicles, the damage per *average* ton of diesel used in a country, when power plants, large industries, long-distance transport, agriculture, and the like are included, could be close to (or below) that for fuel oil and lower than that for gasoline. The exact level of this

average damage will vary from country to country and would need to be validated by an adequate data set.

The main policy implications of this analysis can be summarized as follows. Fuel prices affect the levels of emissions by influencing aggregated or fuel-specific demand. The high environmental costs of fossil fuels highlight the substantial social gains, mainly in public health, that could be achieved by eliminating price distortions caused by subsidies, protection of domestic oil or coal monopolies, and unbalanced taxation of similar products with little regard for their true social costs. These damage estimates thus reinforce the case for prudent economic and fiscal policies. They also show the magnitude of social losses from the current patterns of fuel use in developing countries, which warrants greater attention by policymakers. Finally, the analysis demonstrates the complexity of relationships among various types of damages, fuels, and pollution sources and highlights the need for a thorough investigation of the proper policy response.

The design of appropriate policies should take into account the variety of instruments—both command-and-control and incentive based—that can be employed for internalizing air pollution externalities. Fuel taxes are one possible tool, but many other measures are available and are often more efficient (see, for example, Eskeland and Devarajan 1995). For efficient mitigation of environmental damages from fuel use, an elaborated mix of policy instruments in which sound fuel pricing is complemented with environmental regulations and more targeted incentives needs to be developed. Box 1.1 illustrates this discussion with an example of diesel pricing in South Asia. The main challenge is to find the right mix for each specific case, given the prevalence of particular fuels and sources in a city and country, the existing distortions in fuel

Box 1.1
Urban Air Pollution and Petroleum Pricing in South Asia

As a result of tax differentiation or other fiscal measures, the retail price of diesel is typically lower than that of gasoline worldwide. Industrial countries have narrowed the gap between the prices of these two products, but the differential remains large in many developing countries, especially in South and Southeast Asia (World Bank 1996, 1997f). Whereas the price of gasoline falls in the range typical for OECD countries, retail prices of diesel are kept at levels close to or sometimes lower than import parity prices for this product. In Bangladesh retail prices (in taka, with 51 takas = 1 U.S. dollar) in 1999 were 21.00 per liter 80 research octane number (RON) gasoline, 23.00 per liter 95 RON gasoline, and 12.95 per liter kerosene and diesel. In Pakistan the mid-1999 retail prices (in Pakistan rupees, with 51.89 rupees = 1 U.S. dollar) were 22.19 per liter 80 RON gasoline, 9.44 per liter kerosene, and 9.66 per liter high-speed diesel. An increase in prices of petroleum products in late 1999 widened this gap further. The ratio of gasoline to diesel consumed in Pakistan's transport sector in fiscal 1996–97 was 4.5. Similar trends are observed in India and Sri Lanka. By comparison, the ratio is slightly over 1 in most OECD countries. Gasoline has been historically priced much higher than diesel in South Asia (except in Bangladesh between 1990 and 1997) because it is seen as a fuel used primarily by those who are well off enough to buy vehicles.

As a result of the difference between the retail prices of gasoline and diesel, vehicle technology in South Asia is increasingly focusing on small diesel-engine vehicles such as passenger cars and, recently, three-wheel taxis. These vehicles are more costly to purchase than those powered by gasoline engines, but they may be more economical in the long run after taking into account the higher fuel economy of diesel and its substantially lower retail cost. This increasing dieselization of urban transport is environmentally costly because of the higher levels of toxic emissions emitted by diesel engines and the consequent health damages. Furthermore, a

(continued)

Box 1.1 *(continued)*

Urban Air Pollution and Petroleum Pricing in South Asia

massive switch away from gasoline undermines the fiscal objective of this price regime and creates distortions in the supply chain.

What would be the appropriate policy response to the adverse environmental and health impacts of dieselization of vehicles in South Asia? Several issues need to be considered.

How diesel is used. From the point of view of the public health impact, what is important is the amount of diesel used in urban transport. The environmental impact of diesel used in intercity transport (for example, long-distance trucking and railways), large industries, power, and agriculture is not as much of a concern. The choice of policies will depend on the breakdown in the use of diesel across economic sectors and within the transport sector.

Options for reducing the economic incentives for switching to diesel. There are three possibilities: (a) decrease the retail price differential between gasoline and diesel; (b) make owning a diesel vehicle more expensive—for example, by taxing light-duty diesel vehicles much more heavily than vehicles using cleaner fuels; or (c) use a combination of these two methods. Alternative (a) affects all diesel, including that used in long-distance transport, community generators, farming, and the like. Thus, if this measure is employed, it should be motivated by broad fiscal and macroeconomic considerations, although urban air pollution problems can reinforce the case. Alternative (b), depending on the design of the fiscal measures employed, can be used to target light diesel vehicles directly. A combination of the two measures (alternative c) can help to harmonize the fiscal and environmental objectives.

A tax on diesel. In South Asian countries it is imperative, from both the fiscal and the environmental perspectives, to abolish diesel subsidies and impose a reasonable level of taxes on diesel fuel (for revenue raising, recovery of road user costs, and so on). But should taxation of diesel incorporate the specific objective of reducing urban air pollution? Environmental costs could provide social justification for an "extra" tax on diesel used in urban transport. This level of tax is not, however, justifiable for other, less damaging uses of diesel. Unless it is possible to discriminate between diesel users on the basis of their environmental performance (through tax rebates or other mechanisms), a high environmental tax on diesel will penalize the large numbers of rural poor for sake of mitigating urban air pollution problems that essentially affect only the relatively small share of people living in the largest, highly motorized cities. An alternative approach to reversing the dieselization of light urban vehicles could be a skillful combination of (a) increases in diesel prices (to the import parity level plus a moderate tax) and (b) a set of other measures targeted at reducing the use of diesel by urban vehicles and at improving urban air quality, such as a higher tax on light-duty diesel vehicles and enforcement of fuel and emissions standards. In this context, liberalized diesel prices driven by sound economic and fiscal policies, even if they fall short of incorporating the social costs of urban air pollution, constitute a powerful tool for environmental improvement.

The price of gasoline. It is worth remembering that vehicle dieselization is encouraged by the combined effect of a low price for diesel and a high tax on gasoline. The high gasoline tax, the low profit margin fixed by governments for the sale of gasoline, and low kerosene prices have together led to another type of adverse environmental behavior: the adulteration of gasoline with kerosene, which results in higher vehicular emissions. This further highlights the importance of balanced taxation of petroleum products that are close substitutes. Many governments, including those of OECD countries, find high gasoline taxes an attractive measure for increasing budget revenues. OECD countries, however, keep the gap between gasoline and diesel prices narrow or use other enforcement mechanisms to prevent undesirable fuel substitution. In South Asia the wide use of diesel and kerosene by various segments of the population, including the urban and rural poor, prevents the imposition of high taxes on these products. There, tax policies for gasoline should take into account the opportunities for substitution among fuels, both in the long term (switching to diesel cars) and in the short term (adulterating gasoline with kerosene; switching to liquefied petroleum gas). These opportunities are particularly large in the absence of a strong regulatory framework. As a result, a discord between tax policies for gasoline and for other

(continued)

Box 1.1 *(continued)*
Urban Air Pollution and Petroleum Pricing in South Asia

petroleum products can unleash interfuel substitution that interferes with the revenue-raising objective of a gasoline tax. In these circumstances, increases in the price of gasoline that make it more difficult to reduce the price gap between gasoline and diesel do not seem warranted.

Prospects for strengthening the regulatory framework. The introduction and enforcement of vehicular emissions and fuel standards can correct for the lack of market incentives for cleaner fuels and vehicles. Reducing perverse price incentives remains crucial; the stronger are the perverse incentives, the more difficult it is to enforce environmental regulations. Fuel standards can themselves influence more balanced pricing. (For example, tighter quality standards for automotive diesel imply higher production costs and prices.) In OECD countries a combination of effectively enforced standards and the reinforcing impact of fuel pricing has dramatically improved the environmental performance of vehicles over the past two decades. A similar twofold approach is needed if the alarming air pollution situation in South Asian cities is to be reversed.

Clean diesel technology. With the advent of new technologies such as continuously regenerating traps, state-of-the-art diesel vehicles that use ultra-low-sulfur diesel fuel can be as clean as vehicles that use compressed natural gas. It is important to recognize that reducing the use of diesel is not the only way to address vehicular pollution and carry out a cost-effective strategy for reducing toxic emissions from urban transport.

markets, and the capacity of governmental institutions. Meeting this challenge will require serious analytical work that integrates the damage estimates with assessments of mitigation options and the impact of alternative policy measures.

Summary of Findings

The major qualitative findings of this exercise are as follows:

- The environmental costs of fuel use in large developing-country cities can be so high that marginal damage costs may exceed both producer and retail prices for some fuel uses.
- In highly polluted urban areas, local health effects dominate the damage costs from fuel use, with global climate change impacts being far less significant.
- Vehicles and small stoves and boilers are responsible for most of the health and overall damages from fuel use, while large sources contribute the most to climate change impacts. This implies that the overlap between measures for addressing local and global issues is likely to be limited.
- The sectoral differentiation in fuel use is at least as significant for the environmental costs of fuel combustion as the differences in type of fossil fuel used.
- Marginal damage costs per ton of "local" pollutants vary greatly across sources and locations. They are much higher for small (low-stack or low-level) sources because of dispersion and exposure patterns.
- Diesel-powered urban vehicles and small stoves and boilers that burn coal, wood, or heavy oil impose the highest social costs per ton of fuel. The greatest disparities between local and global damage costs are also found for these fuel uses.
- The large range of environmental damages for different combinations of fuels, sources, and locations limits the efficacy of simple fuel-pricing measures and requires a skillful mix of policy instruments able to send highly differentiated signals to various users of fuels.

2 From Fuel Use to Exposure Levels

The primary aim of this paper is to assess the major air pollution problems and tradeoffs in several large urban conurbations and to draw broad qualitative conclusions. The study covers six cities, five fuels (coal, fuel oil, automotive diesel, gasoline, and wood), and five sectors: power; district heating, large industry and commerce; small industry and commercial premises; households; and urban transport (vehicles). The interaction of these sets yields twelve fuel uses (see informal table in Chapter 1, p. 7).

Emissions Inventory

For each fuel use, the model employs a set of standard emissions factors for particulates (total suspended particulates/PM_{10}), sulfur dioxide (SO_2), and nitrogen oxides (NO_x), compiled from WHO and USEPA documents (USEPA 1986; WHO 1989). The factors are given in Annex A. Emissions factors for particulates from small fuel oil and diesel uses are chosen from an upper end of estimates, using the assumption that in developing countries the equipment (boilers, stoves, or vehicle engines) is not up to the current technology standards in industrial countries and is not well maintained.[6] When local emissions factors are known, default data can be refined.

In most cases, emissions factors, especially for particulates and SO_2, depend on the quality of fuel and the level of controls typical for pollution sources within a sector. The latter point is especially relevant for power plants and large industries, where controls on particulates are relatively inexpensive and have been widely adopted. These parameters are factored into the model. Fuel quality parameters include ash and sulfur content of coal and sulfur content of petroleum products. The model therefore estimates not only the emissions that correspond to the current quality of fuel used and the existing level of controls but also alternative volumes of emissions for a variety of available control measures, such as improvement in fuel quality, fuel switching, or technological change. This makes it possible to calculate the physical impact and the economic benefits of alternative abatement measures in different sectors.

Modeling Atmospheric Dispersion

The next step in apportioning responsibility for local damages associated with poor air quality to particular sectors or fuels is to establish a link between the absolute change in ambient concentrations of pollution and the unit change in emissions from each sector.

The dispersion patterns for emissions from different sectors are very different. One of the most sensitive parameters is the height of the emissions stack. Maximum concentrations typically occur at a distance of about 10 times stack height. Low-stack emissions have the greatest impact on ambient levels of pollution in the immediate proximity of the pollution source. High-stack emissions are dispersed over large areas and contribute far less per unit

of emissions to ground-level concentrations. This explains why emissions of pollutants in industrial areas can be higher than those experienced in residential areas but the concentrations can be lower (as is seen in monitored SO_2 levels in Shanghai).

For this rapid assessment exercise, a simple dispersion model was chosen that assigns different dispersion patterns to high-stack (over 75 meters), medium-stack (25 meters–75 meters) and low-stack sources (less than 25 meters). High-stack sources are synonymous with modern power plants; medium-stack sources with large industrial plants, district heating plants, and suboptimal power generators; and low-stack, or low-level, sources with small industrial and commercial users, transport, and the domestic sector. Thus, the grouping of pollution sources by five sectors in this study is motivated by two principal considerations: (a) differentiation of major economic sectors by fuel use and emissions patterns (for example, transport versus industry) and (b) differentiation of pollution sources by the typical height of the emissions stack, as required for dispersion modeling (for example, large power plants versus smaller district heating stations, or large versus small industry). Because of the great sensitivity of the dispersion model to the level (high, medium, or low) of pollution sources, the main criterion for defining an industrial activity as large or small in this analysis is the height of the emissions stack.

The dispersion model computes annual-average and spatial-average concentrations of relatively stable air pollutants. (See Annex B for model equations.) It does not take account of photochemical reactions and secondary pollution formation in the ambient air. Although the simulation is in many respects simplistic, it is reasonable for the purposes of rapid assessment and offers a better alternative to the assumption that each ton of emissions of a pollutant from any sector in an area makes an equal contribution to concentration of the pollutant.

Secondary particulates

Most of the health effects from fuel combustion are associated with exposure to particulates, especially to *fine particles* of less than 2.5 microns in aerodynamic diameter (see Chapter 3). The model estimates average concentrations resulting from direct emissions of particulates from fuel burning. However, two other main pollutants emitted in the process of fuel burning and assessed in the study—SO_2 and NO_x—also contribute to ambient levels of fine particulates, forming secondary *sulfates* and *nitrates*.

Empirical measurements show that the proportion of fine particulates formed by sulfates and nitrates may vary greatly—typically, from 10 to 50 percent for sulfates and from 10 to 40 percent for nitrates in U.S. cities.[7] Sulfates are a function of the sulfur content of fuels and have a significant presence in the air in areas where high-sulfur coal or heavy oil is widely used. Nitrates are most significant in cities where PM_{10} and SO_2 emissions are effectively controlled and cars using high specifications of gasoline are major polluters. In Bangkok and in Chongqing (China) organic carbon compounds account for a large fraction of fine particulates, reflecting the role of emissions from diesel and two-stroke vehicles in Bangkok and of smoke from combustion of coal in Chongqing. In Chongqing sulfates also represent a substantial share of fine particulates. Care is required, however, in making generalizations about the relationship between sulfates and fine particulates, since the sources and species characteristics of fine particulates can vary so much across locations. Available evidence suggests that the share of nitrates in the cities in our sample is likely to be at the lower end of the U.S. range (10 percent or less to 20 percent).

To ignore the contribution of SO_2 and NO_x to the levels of ambient particulates would significantly understate the social costs of fuels, as well as the benefits of measures that control emissions of these compounds. In this exercise we have opted for a very simple approach to estimating the conversion of SO_2 and NO_x into sulfates and nitrates (see Annex B for details). Overall, the introduction of secondary particulates increases the local damage costs of fuels by about 25 percent in this analysis, which seems to be a plausible estimate. Given that uncertainty is greater about this portion of the damage costs than about that related to the direct dispersion of PM_{10} emissions, it is worth noting that a 25 percent difference in the local environmental costs of fuels does not change any of the qualitative findings or conclusions of Chapter 1, which are very robust.

From concentration to exposure

The computed concentration levels are still a very crude approximation of actual exposure, since people are not evenly located over a city area, and they spend different amounts of time in different areas of the city. The relative proximity of certain sectors to population centers is an important factor. For example, within a central urban area, concentrations of pollution may be formed almost wholly by emissions from road transport, and in many residential areas, by emissions from households and small businesses. Larger industries may also affect certain residential areas, depending on urban zoning. The dispersion model does not simulate the configuration of dispersion patterns and peak concentrations from industrial zones or individual point sources (for which the Gaussian plume dispersion model would be more appropriate). It simply assumes that the aggregated emissions from large sources come from a stack in the center of the city and are dispersed equally in all directions. Therefore, depending on whether the impact from large point sources falls on areas with higher or lower population density compared with the city average, the exposure levels attributed to these sources may be understated or overstated.

The dispersion model that has been adopted gives different weights to different sectors according to their contribution to annual mean concentration levels averaged over the entire city or agglomeration. For exposure assessment, however, these weights should instead reflect the contribution of sectors to the places where most people spend most of their time. Such analysis would require much more information and more sophisticated dispersion modeling. Because of the great variety of counterbalancing factors, the overall impact would be uncertain and is unlikely to be drastically different at the high aggregation level used for the current study.

The study generally maintains a reasonable degree of conservatism in estimating environmental damage costs (as will be shown in subsequent chapters). It assumes that many concentration "hot spots" have limited exposure effect and therefore adjusts the level of exposure per "average" resident to 70 percent of the calculated value for the Shanghai agglomeration and 80 percent for the other cities.[8]

Results for the Six Cities

Table 2.1 shows the results of the emissions inventory for the six sample cities. The inventory demonstrates how distinct is the role of different sectors with respect to different pollutants. A dominant share of SO_2 emissions (nearly 90 percent) comes from power plants and large boilers. Road transport is the largest single source of NO_x emissions (over 40 percent), although contributions from power and large industry are significant. The contribution of power plants to PM_{10} emissions appears to be far less because of the standard

use of dust controls. Industrial and commercial boilers are the main sources of PM_{10} emissions.

Table 2.1 highlights the sectoral pattern of emissions, which, as discussed above, may be very different from the exposure pattern. Figure 2.1 illustrates the difference between sectoral contributions to the emissions and exposure levels of PM_{10} for the whole sample of six cities. Whereas large sources—power plants and industries located within the boundaries of a city or agglomeration—contribute 74 percent of emissions, their contribution to the PM_{10} exposure of an average resident of a metropolitan area appears to be only 13 percent. (Some residential areas located near industrial hot spots can experience a far greater impact of industrial emissions, but the given data describe a situation averaged over a large metropolis.) Nearly 90 percent of this exposure—and thus of the related health impacts—comes from small sources such as small-scale industry and commerce,

Table 2.1 Emissions from fuel use, by sector: Six cities (thousands of tons)

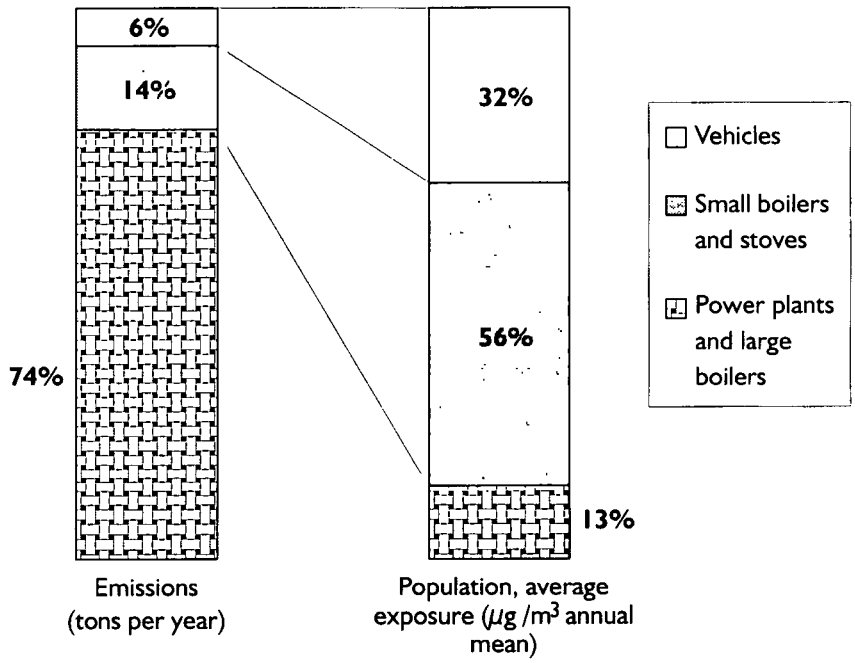
	Mumbai	*Shanghai*	*Manila*	*Bangkok*	*Krakow*	*Santiago*	*All six cities*	*Percentage, by pollutant*
PM_{10} emissions	*23.9*	*409.4*	*23.1*	*16.2*	*44.6*	*7.9*	*525*	100
Power plants	1.9	28.5	1.9	n.d.	32.9	n.d.	65	12
Large industrial and commercial boilers	7.7	297.6	6.9	4.7	6.7	1.3	325	62
Small industrial and commercial boilers	9.1	70.7	2.0	1.0	3.2	2.3	88	17
Households	2.1	6.5	1.0	0.4	0.8	2.1	13	2
Vehicles	3.1	6.1	11.4	10.1	1.0	2.2	34	6
SO_2 emissions	*52.0*	*609.7*	*167.6*	*58.5*	*100.7*	*19.4*	*1,008*	100
Power plants	21.4	218.2	71.9	n.d.	88.7	n.d.	400	40
Large industrial and commercial boilers	17.2	304.9	72.2	46.0	5.5	12.4	458	45
Small industrial and commercial boilers	9.0	45.9	10.4	5.0	3.9	0.0	74	7
Households	1.9	26.7	3.8	1.1	2.0	3.6	39	4
Vehicles	2.5	14.0	9.3	6.4	0.7	3.4	36	4
NO_x emissions	*48.0*	*309.1*	*118.4*	*86.5*	*41.4*	*43.1*	*646*	100
Power plants	11.0	118.6	10.2	0.0	32.5	n.d.	172	27
Large industrial and commercial boilers	6.8	119.0	16.8	16.1	1.3	4.3	164	25
Small industrial and commercial boilers	2.7	13.8	4.7	2.3	0.7	0.6	25	4
Households	1.6	3.5	1.4	0.8	0.1	1.6	9	1
Vehicles	25.8	54.1	85.3	67.4	6.8	36.6	276	43

n.d. No data.
Source: Authors' calculations.

households, and vehicles. These findings are broadly consistent with the more elaborate modeling of air pollution dispersion from various urban sources reported in a number of studies (for example, World Bank 1995a, 1997b; Adamson et al. 1996).

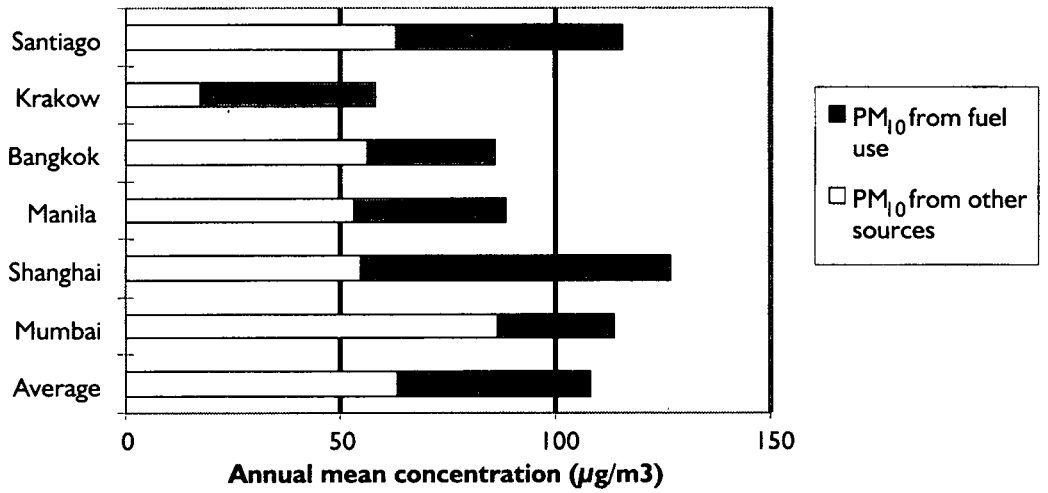
Finally, Figure 2.2 demonstrates how a focus on reducing emissions from fuel combustion can improve urban air quality, as measured by ambient levels of PM_{10}.[9] The improvements can be very substantial, but in all except one of the sample cities (Krakow), these policies and measures will probably not be sufficient to ensure that urban air quality meets pre-1997 WHO recommendations. Abatement of fuel combustion emissions will, however, achieve a greater reduction in air pollution damages than Figure 2.2. might indicate. This is because the particles from fuel burning are believed to be more harmful to human health than particles of a different nature that are present in urban air (see Chapter 3).[10]

Figure 2.1 Sectoral contributions to the emissions and exposure levels of PM_{10} from fuel use: Six cities, 1993 (percent)



Note: The ambient concentrations of PM_{10} are entirely attributed to emissions from fuel combustion, including the secondary impact of SO_2 and NO_x emissions. Only sources burning fuels within a city or agglomeration are considered. Long-range pollution from power plants and other high-level sources that are located outside the boundaries of a city is not assessed, although it does contribute to levels of SO_2 (and thus sulfates and PM_{10}) within the city area.
Source: Authors' calculations.

Figure 2.2 Contribution of fuel use to ambient levels of PM_{10} in urban air: Six cities, 1993



Note: The levels of PM_{10} from other sources are defined as differences between the annual mean levels of PM_{10} based on ambient measurements and the estimated contributions from fuel use. Data are for 1993. Note that the difference may have a significant margin of error and should not be used for calculating damages from the "other sources."
Source: Authors' calculations.

3 The Health Effects of Air Pollution

The economic estimates of health and nonhealth damages are based on certain methodological tools and are as credible as these tools are. This chapter and the next two discuss the methodological issues of valuing a variety of environmental impacts. The issues in valuing health impacts fall into two groups: (a) the actual identification and measurement of these impacts and (b) estimation of monetary values for associated morbidity (illness) and mortality (death). This chapter focuses on the first set of issues.

Fuel Combustion and Health

Fuel combustion is responsible for the direct emission and secondary formation of several pollutants known to be damaging to human health (see, for example, Lave and Seskin 1977). The health effects of exposure to particulate matter include additional cases of premature mortality from respiratory illnesses and cardiovascular disease, increased prevalence of chronic bronchitis, and upper and lower respiratory tract infections. Another very toxic pollutant is lead, which is used as an additive to gasoline in many countries. Exposure to atmospheric lead is associated with neurodevelopment effects on children. Other by-products of fuel combustion—sulfur dioxide (SO_2), nitrogen oxides (NO_x), and volatile organic compounds (VOC)—also impose health impacts, either directly (through increased morbidity) or, to a larger extent, by contributing to ambient levels of particulates and ground-level ozone (see Table 3.1). Most of the available studies indicate an effect of ozone

Table 3.1 Linkages between air pollutants and health effects

Pollutants related to fuel combustion observed in the ambient air	*Primary pollutant*	*Secondary pollutant*	*Precursor of secondary pollutants*	*Recognized impact on*	
				Mortality	*Morbidity*
Fine ($PM_{2.5}$) and inhalable (PM_{10}) particles	√	√		√	√
Sulfur dioxide (SO_2)	√		√ ($PM_{2.5}$)		√
Nitrogen oxides (NO_x)	√		√ (ozone and $PM_{2.5}$)		√
Volatile organic compounds (VOCs)	√		√ (ozone and $PM_{2.5}$)		√
Ozone (O_3)		√			√
Atmospheric lead	√				√

Note: Primary pollutants are direct by-products of fuel combustion. Secondary pollutants are formed in the air through chemical reactions
Source: Compiled by authors from WHO air quality guidelines and a review of various studies on air pollution and health.

on respiratory illness and lung function decrements. There is also growing, although still debated, evidence of an impact on mortality rate (Holgate et al. 1999).

Over the past decade, epidemiologists have reexamined the evidence on the links between particulates, SO_2, and health. This work has led to important shifts in emphasis on different air pollutants and underpins the USEPA and European Union proposals to revise their ambient air quality standards for particulates. A brief review of the current state of knowledge from the perspective of economic analysis of the benefits of reducing air pollution follows. More information is provided by USEPA (1997).

Coarse and fine particulates

As monitoring methods and data analysis have become more sophisticated, the focus of attention has shifted gradually from total suspended particulates (TSP) to inhalable particles below 10 microns in diameter (usually measured as PM_{10}) and to fine particles below 2.5 microns ($PM_{2.5}$). Most fine particles measured as $PM_{2.5}$ are smaller than 1 micron (PM_1). In this paper TSP larger than 2.5 microns is termed coarse particles. TSP was the most common measure of particulates until the 1980s, although some countries used "black smoke," which approximates quite well to PM_{10}. The USEPA shifted to a PM_{10} standard in 1986 after it became apparent that high levels of TSP were often the result of wind-blown dust rather than of pollution and had little impact on human health (see, for example, Ozkaynak and Thurston 1987). Subsequently, it was realized that even PM_{10} may contain substantial fractions of wind-blown dust, as illustrated by the fact that many of the places in the United States that exceed the PM_{10} standard are dry, thinly populated areas.

Evidence from studies completed in the past decade suggests that fine particulates are most likely responsible for the excess mortality and morbidity associated with high levels of exposure to particulates. Since most studies have used PM_{10} rather than $PM_{2.5}$ as their exposure metric (simply because $PM_{2.5}$ has not been routinely monitored to date), this conclusion is based on several indirect but compelling facts. First, in most of the epidemiological studies finding associations between PM_{10} and adverse health effects, there is a high correlation between PM_{10} and $PM_{2.5}$ and a low correlation between PM_{10} and coarse particles. Second, $PM_{2.5}$ tends to penetrate indoors at a much higher rate than do coarse particles. Third, fine particles penetrate deeper into the lung and are likely to be more reactive there.[11] Although the weight of evidence indicates that there should be greater concern about fine particles, a potential effect from PM_{10} particles greater than 2.5 microns cannot be completely ruled out.

In light of this new evidence, the USEPA promulgated a new U.S. ambient standard for fine particulates: an annual average of 15 $\mu g/m^3$. This would represent a substantial tightening of the current standard of 50 $\mu g/m^3$ for PM_{10}, since, typically, fine particulates account for around 60 percent of PM_{10} (although the ratio varies substantially across locations). The European Union has proposed tightening the PM_{10} standard to 30 $\mu g/m^3$ by 2005 and to 20 $\mu g/m^3$ by 2010—effectively, even stricter than the U.S. standard.

Almost all fine particulates are produced, directly or indirectly, as a result of burning fuels. Industrial and other processes that produce large amounts of dust—such as cement manufacturing, mining, stone crushing, and flour milling—tend to generate particles larger than 2.5 microns. The species composition of fine particulates varies considerably across locations.

Exposure to sulfur dioxide

It is now generally agreed that for low to moderate levels of exposure to SO_2, the primary health risks are to asthmatics and, in particular, to asthmatics taking exercise outdoors. High levels of SO_2 can provoke breathing difficulties and even severe asthma attacks among this group. Exercise seems to make people more vulnerable because the person exercising breaths through the mouth rather than through the nose, and more SO_2 penetrates deep into the lungs. The effect is reversed as soon as exposure levels fall, and the evidence suggests that any permanent damage is slight or not observable. Animal experiments at very high levels of exposure—more than 500 $\mu g/m^3$—show that prolonged exposure can produce temporary bronchitis, but, again, this is reversed once exposure levels are reduced. Furthermore, there are no signs of heart arrhythmia of the kind associated with fine particulates, which is the primary mechanism implicated in premature mortality from air pollution.

In a few towns and cities in the world, the population is exposed to annual average SO_2 levels higher than 200 $\mu g/m^3$. Chongqing, China, is the largest such urban center. There, it is the use of poor-quality coal for residential heating and small-scale boilers that is the primary source of the SO_2; residential areas have SO_2 levels as high as or higher than those in industrial areas, and the highest levels are observed in winter. A similar pattern can be observed in other cities with relatively high annual average SO_2 levels—Istanbul and Katowice today, or Leipzig and Prague in the past. Little is known about the effects of such exposure because it is virtually impossible to disentangle the effects of high SO_2 from those of high levels of particulates, also observed in those areas. Animal experiments suggest that the main effects might be an increased incidence of respiratory disease, but this need not be accompanied by a significant increase in mortality.

Aerosol acidity

At one time it was believed that acid aerosols were the major cause of ill health among those exposed to air pollution. However, recent studies have found effects from particulate matter even in areas with very low acidity. Only a few studies have demonstrated an independent effect of sulfuric acid aerosols or particles coated with hydrogen ions. Careful monitoring has shown that even high levels of SO_2 need not be associated with high aerosol acidity (aerosols in Chongqing are almost neutral) because there are two quite separate processes by which SO_2 is converted to aerosols: photochemical reactions that produce sulfuric acid, and atmospheric reactions with ammonia to produce ammonium sulfate. In the eastern United States, peak levels of SO_2 emissions occur in the summer, when conditions are favorable for photochemical reactions. In China and Eastern Europe peak emissions occur in winter, making atmospheric reactions with ammonia more important. Although ambient acidic particles may play a role in lung inflammation and subsequent adverse health outcomes, the current evidence indicates that acidity is not necessary to mediate particle-associated health effects.

This review suggests that a focus on small particulates (PM_{10} and smaller) should provide a good indication of the health effects from fuel combustion.

Air Pollution Dose-Response Studies

Dose-response studies involve estimating physical or medical relationships linking

environmental ambient concentrations of air pollutants with mortality and morbidity outcomes for susceptible population groups. Ultimately, the question of whether air pollution causes a deterioration of human health must be explained by scientific theory. However, consistent significant statistical relationships between concentrations of a particular pollutant and excess mortality or other health impacts, found under a variety of circumstances, is taken as a plausible basis for causality, given the uncertainty inherent in research on human biology. Where the biological pathway by which the pollutant affects human mortality and morbidity is unresolved, consistent findings of multiple epidemiological studies are typically taken as "proof" of causality. More specifically, the absence of a theory linking a pollutant to a particular health outcome cannot nullify the observation (although an inability to replicate the findings on other data might). Once the statistical relationship has been uncovered, it can be used to predict the number of health events attributable to poor air quality in different contexts, based on the size of the population at risk and the concentrations of pollutants to which this population is exposed.

The most accurate way of measuring the health impacts of air pollution in a given area is to conduct epidemiological studies to establish dose-response relationships linking environmental variables to observable health effects. However, given the time and cost involved in such studies (as well as the likelihood of encountering problems of data availability), dose-response relationships established in other locations may often have to be used instead. The availability of other research results can be used to shrink the uncertainty associated with individual studies. Although a single study that finds a statistically significant association between a health effect and a specific air pollutant does not prove causality, the inference of causation is strengthened if (a) epidemiological results are duplicated across several studies; (b) a range of effects is found for a given pollutant; and (c) these results are supported by human clinical and animal toxicology literature.

An approach to reducing the uncertainty associated with individual studies is to use meta-analytical techniques that produce a "best estimate" in which more confidence may be placed. Meta-analysis is a generic term for the statistical pooling of results from several studies to obtain aggregate values that are more reliable. The meta-analytical approach recognizes the inherently stochastic properties of the estimation process: repeated identical studies will lead to different results because each study is a sample drawn from a distribution of possible studies. It is the mean and variance of this "mother" distribution that meta-analysis seeks to estimate.

Meta-analysis therefore assumes that each sample has the same underlying dose-response relation. Individual studies have to be tested for this assumption. Pooling is usually carried out only when there is no significant difference between the separate sample estimates. For the outcome of meta-analysis to be of good quality, it is important to ensure that pooling does not include studies in which essential variables (covariates) have been omitted.

In the air pollution literature, subjective judgment by experts has been also used to pick the most appropriate study or studies. This approach seeks to determine in a less formal way whether there are significant quality or data differences among the studies or whether certain studies are more relevant to the study area (for example, because of similar pollution concentrations, copollutants, or background demographics).

A number of meta-analytical reviews that have been presented in the literature (Ostro 1994,

1996; Pope and Dockery 1994; Schwartz 1994a) can serve as the basis for extrapolating dose-response relationships to situations for which no specific epidemiological studies have been done. Further development of this approach would benefit from putting more effort into checking systematically omitted variables through statistical tests and identifying all the important covariates to be used in the extrapolation.

Studies of the mortality effects of air pollution are often conducted using Poisson regression techniques. Poisson regression assumes that the number of deaths or other health impacts follows a Poisson distribution (as an approximation to the binomial distribution). The coefficients of the Poisson regression can typically be interpreted as the proportionate change in the number of deaths per unit change in the level of the pollutant. Following the practice often used for meta-analysis, each study estimate is weighted by the inverse of the variance associated with the study's regression coefficient.

The coefficient used here attempts to reflect our assessment of the best estimate taken from a carefully selected pool of studies, given a wide range of sensitivity analyses involving alternative lags, model specifications, overdispersion, outliers, and alternative methods for controlling for seasonality and other potential confounders. Usually (but in not all cases), judgments on the models are not simply the highest effect estimates but are based on the strongest association (highest *t*-statistic), the best model fit, or consideration of residuals.

Over the past decade, more than two dozen epidemiological studies have indicated an association between mortality and particulate matter. Chronic exposure to particulates can lead to premature death by exacerbating respiratory illness, pulmonary disease, or cardiovascular disease. Acute exposure (short-term peaks in the level of particulates) can increase the chance that a person in a weakened state or an especially susceptible person will die. Some studies (for example, Katsouyanni et al. 1997; Holgate et al. 1999) found correlations between mortality and other pollutants such as SO_2 or ozone. Most studies use single-pollutant rather than multipollutant regressions. To the degree that different air pollutants tend to be correlated over time, this procedure might mean that different pollutants are used to explain what are essentially the same deaths several times over. Using estimates for different pollutants from these studies may therefore lead to substantial double-counting. However, multipollutant regressions may make the interpretation of the results even more difficult (see, for example, Schwartz et al. 1996). In the future, statistical techniques for pooling multipollutant studies would be a valuable extension of standard meta-analysis for a single pollutant.

In this study we provide quantitative estimates of mortality effects related to particulate matter only. That pollutant then serves as a potential index for many correlated pollutants. It is important to note that the effects of particulates on mortality have been observed in areas with both high and low SO_2 and ozone concentrations, in areas where particulates peak in the summer months, and in areas where the peaks occur in winter. These diverse effects of particulate matter have been reported by the largest number of epidemiological studies for any pollutant and are confirmed by toxicological studies. It therefore appears reasonable to attribute mortality effects to changes in particulate concentrations.

Application to Developing Countries

Most dose-response studies have been conducted in industrial countries. This raises

the question of whether extrapolation of the results to developing countries is valid. Although some uncertainty remains, recent studies undertaken in developing-country cities such as Bangkok, Mexico City, and Santiago lend support to extrapolation. In addition, epidemiological studies typically provide information on the percentage change in mortality attributable to an absolute change in ambient particulate matter. This may make the studies more appropriate for extrapolation, since they predict changes in relation to the baseline mortality rate, which may differ greatly between study areas. To determine the number of excess deaths due to exposure to higher concentrations of particulates, or the number of deaths prevented as a result of lower concentrations, the percentage change and the baseline rate in the affected area must be determined.

When the results from dose-response studies of air pollution in industrial countries are being applied to developing countries, four issues should be carefully addressed:

1. *Measures of particulate matter:* the availability of data on PM_{10} and $PM_{2.5}$ in both the original epidemiological study and the country in question.

PM_{10} is a better proxy than TSP for fine particulates and is employed in a variety of recent studies. A number of meta-analytical estimates of changes in mortality have been produced by applying standard conversion factors across dose-response studies that use different measures of particulate matter: TSP, BS (black smoke), or coefficient of haze (COH). These estimates, however, are less reliable than those for PM_{10} because variations in levels of TSP or other measures of particulates may be quite different from those for PM_{10} and even more different for $PM_{2.5}$, especially in places with high levels of road or wind-blown dust, such as many cities in developing countries. Estimates based on studies that explicitly measure PM_{10} can significantly reduce the uncertainty involved in converting from one measure of particulate matter to another. Therefore, studies using direct measurements of PM_{10} (or $PM_{2.5}$) have been given primary attention in this analysis. To the extent that additional epidemiological studies can be undertaken, efforts should be made to put in place and utilize monitors that measure either PM_{10} or $PM_{2.5}$.

2. *The existing pollution concentration.* In most of the available studies, PM_{10} has a mean concentration of about 50 to 60 $\mu g/m^3$, with maximum values of about 150 to 200 $\mu g/m^3$, and consists largely of particles generated by combustion processes. Caution should be exercised in extrapolating these air pollution and mortality results to areas where the concentration or mix of pollution may be different. For example, in those cities in the developing world where annual mean concentrations of TSP exceed 300 $\mu g/m^3$ and these high levels cannot be easily explained by the pattern of fuel use, it is likely that a substantial portion of the TSP consists of particles larger than 10 microns and that the highest concentrations are driven by coarse, geologic particles. (Calcutta and Delhi, in India, illustrate this situation.) A direct extrapolation from the available studies over the whole range of concentrations may be misleading. This is well shown by Cropper et al. (1997), which found that in Delhi the change in mortality risk per unit change in TSP concentration is significantly lower than in the United States (see Table 3.2). In these cases, one option would be to apply the dose-response functions from industrial countries only to a concentration up to a certain limit; say, of 200 $\mu g/m^3$ TSP for Delhi. Another option is to posit a nonlinear function that begins to level off at some upper cutoff point.

The need for more studies in developing countries, especially those with very different

pollution mixes and exposure patterns, is obvious. In this study, however, we analyze only the impact of incremental PM_{10} concentrations resulting from fuel burning. As Figure 2.2, above, demonstrates, these incremental values fall within a typical range of concentrations in most of the dose-response studies. This reduces the uncertainty of applying the results of existing studies, based on PM_{10}, to different pollution mixes and concentration levels.

3. *Disease-specific mortality profile.* In some cases the distribution of deaths by cause may differ significantly between the country of interest and the country where the original study was conducted. Then, the use of dose-response functions for disease-specific mortality (as opposed to total mortality) or adjustment for this difference may be warranted to improve the accuracy of the projections. For instance, exposure to particulates primarily affects respiratory and cardiovascular deaths, which account for half of all deaths in the United States. In Delhi fewer than 20 percent of all deaths is attributable to these causes. Thus, even an identical reaction by susceptible population groups in Delhi and in the United States to changes in the levels of particulates could result in a lower effect on total mortality in Delhi (Cropper and Simon 1996). The use of dose-response estimates for respiratory and cardiovascular mortality and the associated local disease-specific mortality rates may therefore yield better estimates of the effect of air pollution on mortality, since it better incorporates the local population at risk. Table 3.2 summarizes results for those studies, in the United States and elsewhere, that have considered disease-specific mortality.

There are, however, some advantages in generating estimates using total mortality. The method ensures that, on the basis of the original studies, all mortality cases affected by air pollution are included in the dose-response function. The use of the disease-specific approach in developing countries is often complicated by limited access to disease-specific mortality data and by deficiencies in the death reporting system that may provide distorted information on the actual causes of mortality. When this is the case, the use of all-cause mortality estimates may be preferred. If only cardiovascular- and respiratory-specific mortality causes are used in the dose-response function, the mortality effect may be underestimated if the death certificates used in the original studies were not always accurate or if baseline rates in the country under study are incorrect. Finally, the all-cause mortality approach is more suitable for rapid assessment and cross-country comparisons.

Table 3.2 indicates that the percentage change for total mortality is fairly consistent among the cities, even when results from developing countries are considered, except for the TSP-based Delhi study. The percentage changes for both cardiovascular and respiratory mortality show a larger range across cities than the change for total mortality.

Note that if total mortality functions are used, differences in population characteristics per se, such as age structure, nutritional and overall health status, and smoking rates, and differences in local geography and climate may not necessarily result in bias, since these factors will be reflected in the crude mortality rate. For example, in Chile, where the crude mortality rate is much lower than in the United States, the percentage increase in mortality per $\mu g/m^3$ of PM_{10} is similar to that found in many U.S. cities.

4. *The age pattern of deaths due to air pollution causes.* The age profile of those affected by air pollution may be very different in developing countries than in industrial countries. Although peak effects were observed among

Table 3.2 Disease-specific mortality in selected locations (percentage change per 10 $\mu g/m^3$)

City	Study	Mortality		
		Total	*Cardiovascular*	*Respiratory*
Santa Clara, Calif.	Fairley (1990)[a]	0.8	0.8	3.5
Philadelphia, Pa.	Schwartz and Dockery (1992a)[a]	1.2	1.7	3.3
Utah Valley	Pope, Schwartz, and Ransom (1992)	1.5	1.8	3.7
Birmingham, Ala.	Schwartz (1993)	1.0	1.6	1.5
Steubenville, Ohio	Schwartz and Dockery(1992b)	1.1	1.5	n.d.
Beijing	Xu et al. (1994)[a]	0.7	1.45[b]	6.9[c]
Chicago, Ill.	Ito and Thurston (1996); Styer et al. (1995)	0.6	0.4	1.4
Santiago	Ostro (1996)	1.0	0.8	1.3
Mexico City	Borja-Aburto et al. (1997)[a]	1.0	0.92	1.65
Delhi	Cropper et al. (1997)[a]	0.4	0.78	0.56
Bangkok	Ostro et al. (1998)	1.0	1.4	5.2

n.d. No data.
a. Estimates are converted from TSP using a ratio of 0.55.
b. Pulmonary heart disease.
c. Chronic obstructive pulmonary disease.
Source: Compiled by Bart Ostro.

people age 65 and older in Philadelphia (Schwartz and Dockery 1992a), in Delhi peak effects were reported in the 15–44 age group. This implies more life-years lost as the result of a death associated with air pollution (Cropper et al. 1997). The Cropper study also shows that although the change in mortality per 10 $\mu g/m^3$ change in TSP was lower in Delhi than in the United State, the number of life-years lost in the exposed population of equal size appeared to be similar. This finding has important implications for valuation of the mortality costs that are discussed in Chapter 4.

Estimates for Mortality

The epidemiological studies involve two principal study designs: time-series and long-term exposure studies, which are used to develop quantitative estimates for all-cause mortality associated with air pollution. Time-series studies, which are more common, capture the acute effects of exposure to pollutants. Long-term studies—cross-sectional and prospective cohort studies—yield estimates of the impact of longer-term exposure and indicate both acute and chronic effects.

Time-series studies

Time-series studies correlate daily variations in air pollution with variations in counts of daily mortality in a given city and primarily measure the effects of acute exposure to air pollution. Their advantage is that they do not have to control for a large number of confounding factors, since the population characteristics (age, smoking, occupational exposure, health habits, and so on) in the panel data are basically unchanged. Most of these studies control for time-varying parameters such as weather (temperature, humidity, and precipitation), season, day of week, and presence of other pollutants. A disadvantage is that although the characteristics of the population being studied are fixed, they may nonetheless be important in shaping the slopes

of the dose-response functions. Consequently, these dose-response functions might not be readily transferable to populations with different characteristics as regards diet, smoking, climate, and so on. However, several reviews of these studies suggest that after the different measures of particulate matter are converted into a common metric, the effects on mortality are very consistent (Ostro 1994; Pope and Dockery 1994; Schwartz 1994a). Furthermore, as a result of progress in the past several years, a sufficient number of studies using PM_{10} as the actual measure of exposure are available for meta-analytical purposes.

Table 3.3 summarizes the evidence for nine PM_{10} studies, two of which were conducted in developing countries.[12] The pooled central estimate for the studies, relative to a 10 $\mu g/m^3$ change in PM_{10}, is 0.84 percent.

Time-series studies only indicate the potential effects of short-term variations in exposure, and this effect is likely to be smaller than that of long-term exposure. Offsetting this is the possibility that time-series studies to a large extent measure a "harvesting effect"—deaths that are merely hastened by a few days, weeks, or months as a result of high ambient concentrations of particulate matter. The balance between chronic and acute deaths and the actual extent of any harvesting effect can be assessed when the impact of long-term exposure is brought into the comparison.

Long-term exposure studies

Cross-sectional studies compare differences in health outcomes across several locations at a selected point or period of time and, in principle, capture both the acute and the chronic effects of air pollution. The use of annual mortality rate data in such studies allows both acute and latent health effects to be revealed. Some portion of the long-term response indicated by cross-sectional studies must correspond to the impact of acute effects unearthed by time-series analysis, and the remainder could be attributed to longer-term latent or chronic effects caused by accumulated exposure to the pollutant. Such studies have consistently found measurably higher mortality rates in U.S. cities with higher average levels of particulate matter. However, many more potential explanatory variables need to be modeled in cross-sectional work: variations between cities in smoking rates, diet, income, local industry, age distribution, and so on. A common concern is whether all these factors are adequately controlled. (See, for example, Evans, Tosteson, and Kinney 1984.)

Table 3.3 Estimated percentage change in mortality associated with a 10 $\mu g/m^3$ change in PM_{10}, based on studies that measured PM_{10}

City	*Study*	*Central estimate*	*Low estimate*	*High estimate*
Birmingham, Ala.	Schwartz (1993)	1.0	0.2	1.5
Utah Valley	Pope, Schwartz, and Ransom (1992)	1.5	0.9	2.1
St. Louis, Mo.	Dockery et al. (1993)	1.5	0.1	2.9
Kingston, Tenn.	Dockery et al. (1993)	1.6	−1.3	4.6
Chicago, Ill.	Ito and Thurston (1996)	0.6	0.1	1.0
Los Angeles, Calif.	Kinney, Kazuhiko, and Thurston (1995)	0.5	0.1	1.1
Santiago	Ostro et al. (1995)	1.0	0.6	1.4
Six cities	Schwartz et al. (1996)	0.8	0.5	1.1
Bangkok	Ostro et al. (1998)	1.0	0.4	1.6
Weighted average		0.84		

Source: Compiled by Bart Ostro.

More confidence is placed in the *prospective cohort* type of study, in which a sample of population is selected and followed over time in each location. Such studies are similar in some respects to ecological cross-sectional studies because the variation in pollution exposure is measured across locations rather than over time. However, they use individual-level data so that other health risk factors can be better taken into account. Specifically, the authors of the two prospective studies conducted to date were able to control for mortality risks associated with differences in body mass, occupational exposure, smoking (past and present), alcohol use, age, and gender. Dockery et al. (1993) studied 8,000 individuals in six U.S. cities over a 15-year period. Pope et al. (1995) published results of a seven-year prospective study based on samples of over 500,000 individuals in 151 U.S. cities. Both studies report a robust and statistically significant association between exposure to particulate matter (measured as PM_{10}, sulfates, or $PM_{2.5}$) and mortality.

To illustrate the effects of chronic exposure, we use the Pope et al. study, which has a larger sample size and lower estimates than the Dockery et al. study. When the empirical results for $PM_{2.5}$ were converted to PM_{10} using a ratio of 0.65, a 10 $\mu g/m^3$ change in PM_{10} was associated with a 4.2 percent change in all-cause mortality.

The chosen value for mortality risk

The question of how to integrate the results of the chronic exposure studies with the meta-analytical estimates from time-series studies is not trivial, and it creates new challenges for valuing health effects. Estimates that are based on the chronic studies or that combine the results of the acute and chronic studies imply a long-term exposure to air pollution and thus cannot be used for assessing the short- or medium-term impacts of an annual change in urban air quality without a significant adjustment.

One approach that may be suggested for determining a quantitative estimate for mortality is to use as the upper bound the estimate of 4.2 percent per 10 $\mu g/m^3$ for chronic exposure. An estimate for the lower bound, taken from the acute studies, is 0.84 percent per 10 $\mu g/m^3$. However, since there are only two cohort studies, we would place less weight on these studies in determining a central estimate. As an example of derivation of a central estimate, we apply subjective weights of 0.67 and 0.33 to the acute and chronic effects, respectively, and obtain a value of 1.94 percent per 10 $\mu g/m^3$ for all-cause mortality. This is only an indicative value, subject to significant uncertainty, and it is used here primarily to show that estimates from time-series studies are conservative. (See also the sensitivity analysis in Chapter 6.)

For estimates of disease-specific mortality from cardiovascular and respiratory causes, the acute exposure studies (except for the Delhi study) imply a change of about 2 percent per 10 $\mu g/m^3$. The Pope et al. study gives a 7.2 percent change from a 10 $\mu g/m^3$ long-term exposure. The central estimate, using a similar approach and the same weighting scheme as for total mortality, is 3.7 percent.

The calculations of the mortality effects attributable to exposure to ambient PM_{10} given in this paper use a value of 0.84 percent per 10 $\mu g/m^3$. Given the balance of evidence across various studies, this should be regarded as a lower-bound estimate for the effect of exposure to the levels of PM_{10} attributed to fuel burning or to a similar pollution mix. Furthermore, Tables 3.2 and 3.3 show that the results from studies for Mexico City and Beijing, even when based on TSP measurements, as well as the PM_{10}-based results for Bangkok and Santiago,

are consistent with evidence from industrial countries.

Estimates for Morbidity

In addition to premature mortality, another severe effect of long-term exposure to particulate matter is *chronic bronchitis.* This disease classification includes a variety of illnesses of different severity that typically involve the need to limit a number of activities, take medications, and visit a doctor regularly and that carry a high risk of hospitalization. Abbey et al. (1991, 1993) found a statistically significant association between long-term exposure to TSP and chronic bronchitis. When converted to PM_{10} equivalence and the annual base, a central change in chronic bronchitis was estimated as $6.12 * 10^{-5}$ per $\mu g/m^3$ PM_{10}. Lower and upper changes in chronic bronchitis (within a 95 percent confidence interval) are $3.06 * 10^5$ and $9.18 * 10^{-5}$ (Ostro 1994). In valuing the health effects, the use of a central change estimate for chronic bronchitis means that this health state dominates the health costs of air pollution, exceeding the social cost of premature mortality (see Chapter 4). Still, the results are based on only one study, and in recognition of the inevitable uncertainty, we use the lower estimate of $3.06*10^{-5}$ in our analysis.

Dose-response functions can also be derived for many *lesser health impacts,* such as respiratory hospital admissions (RHA), cardiovascular hospital admissions (CHA), emergency room visits (ERV), bed disability days (BDD), restricted activity days (RAD), asthma attacks (AA), acute respiratory symptoms, and lower respiratory illness in children (LRI). This study uses a comprehensive meta-analysis of these impacts undertaken by Ostro (1994) and updated in Ostro (1996). Most of the morbidity end-points used in the study relate to particulate matter, with two morbidity effects of exposure to SO_2.

Because there are far fewer dose-response studies for morbidity end-points due to exposure to air pollution than for mortality effects, the available meta-analytical estimates are less robust. (For some health end-points, the morbidity estimates are based on only one or two studies and are not truly meta-analytical.) However, as will be shown below, the morbidity effects account for more than half of the overall burden of the health costs attributable to air pollution. The largest portion of the morbidity costs falls on new cases of chronic bronchitis and respiratory symptoms. More epidemiological studies quantifying these effects are therefore needed, especially in developing countries where urban residents suffer from the highest levels of exposure to particulates.

The epidemiological work is not yet sufficiently advanced to provide robust dose-response functions for all the health effects of particulate matter or other pollutant-health combinations. Some health effects identified for nitrogen dioxide (NO_2) are linked to its peaks rather than to annual average levels and are beyond the scope of this rapid assessment. It is worth noting, however, that these effects are not significant for the overall valuation results, as will be seen in the next chapter.

More significant implications for damage estimates are linked to the decision not to assess the health effects of ozone or lead in this exercise. The ambient measurements of ozone are largely not available, except for Santiago, where ozone levels exceed the national standard on quite a few days per year. The indirect data, such as measured levels of NO_2 or NO_x, indicate that ozone levels are unlikely to be high at the moment in the other cities. Adding ozone would require photochemical modeling techniques that can be developed in further applications of the outlined approach. However, previous studies have indicated that the effects of ozone, relative to particles, are small (Krupnick and Portney 1991).

Table 3.4 Air pollution dose-response functions used in the study per $\mu g/m^3$ change in the annual mean level

Health effects	PM_{10}	SO_2
Mortality (percentage change in the all-cause mortality rate)	0.084	
Chronic bronchitis (per 100,000 adults)	6.12 (**3.06**; 9.18)[a]	
Respiratory hospital admissions (per 100,000 population)	1.2	
Asthma attacks (per 100,000 asthmatics)	3,260	
Emergency room visits (per 100,000 population)	23.54	
Restricted activity days (per 100,000 adults)	5,750	
Lower respiratory illness in children (per 100,000 children)	169	
Respiratory symptoms (per 100,000 adults)	18,300	
Cough days (per 100,000 children)		1.81
Chest discomfort days (per 100,000 adults)		1,000

a. A value of 3.06 (low estimate) was chosen for this exercise.
Sources: Ostro (1994); Ostro et al. (1998).

The effects of lead, by contrast, are likely to be substantial (see, for example, Schwartz 1994b; Lovei 1998). Therefore, the total burden on public health of air pollution associated with fuel use is likely to be understated in this study, particularly for such fuels as leaded gasoline, which can significantly contribute to ambient levels of lead and ozone.[13]

Summary of Health Impacts

The mortality and morbidity effects employed in this paper are summarized in Table 3.4.

Quantification of health effects for a particular area

To calculate the change in health effects associated with a change in a pollutant concentration, the following formulas can be applied:

$$\Delta H_i = b_{ij} * \Delta Aj * P \qquad (3.1)$$

where Δ is "change in"; H_i is health impact i per year; b_{ij} is the slope of the dose-response function of health effect i from exposure to pollutant j per year; P is population exposed to the pollutant; and A_j is ambient concentration of pollutant j.

For mortality, which is expressed as the percentage change in mortality risk per unit increase in pollution, the expected change can be calculated by:

$$\Delta H_i = B * (0.01 * b_{ij}) * \Delta A_j * P \qquad (3.2)$$

where B is the baseline mortality rate (for either total or disease-specific mortality).

Note that:

$$\Delta A_j = \max [0, A_j^1 - \max (A_j^0, S_j)] \quad (3.3)$$

where S_j is the relevant threshold or air quality standard; A_j^0 is the initial (background) concentration of pollutant j; and A_j^1 is the new concentration.

Results for the six cities

Table 3.5 illustrates the health impacts attributed to the combined use of coal, petroleum, and fuelwood that have been assessed for each of the six cities in the study on the basis of the above assumptions on dose-response relationships, exposed populations, atmospheric dispersion, and emissions levels.[14]

According to formula (3.3), health effects were calculated for the entire range of incremental concentrations of PM_{10} and SO_2 attributable to fuel burning *only* when "background" concentrations (attributed to other, nonfuel sources) of these pollutants exceeded annual average values of 20 $\mu g/m^3$ for PM_{10} and 50 $\mu g/m^3$ for SO_2. To further exercise caution and rule out the possibility of overstating the health effects from fuel burning, it was assumed that exposure to PM_{10} and SO_2 below these levels caused no health effect. If "background" concentrations were lower than these "threshold" values, incremental concentrations from combustion sources that were used in dose-response calculations were reduced by the difference between the two values.

For example, in Krakow total annual concentrations are 58 $\mu g/m^3$ for PM_{10} and 65 $\mu g/m^3$ for SO_2. Background concentrations are 16 $\mu g/m^3$ for PM_{10} and 25 $\mu g/m^3$ for SO_2. Fuel use is therefore estimated to contribute 42 $\mu g/m^3$ to PM_{10} levels and 40 $\mu g/m^3$ to SO_2 levels. Table 3.5 shows that health effects are calculated for 38 $\mu g/m^3$ (58 – 20) of PM_{10} and 15 $\mu g/m^3$ (65 – 50) of SO_2. In four of the cities ambient levels of SO_2 are well below 50 $\mu g/m^3$. (The exceptions are Shanghai and Krakow.) In these cases no increase in exposure to SO_2 from fuel burning that would cause adverse health

Table 3.5 Health effects from fuel combustion: Six cities

	Mumbai	*Shanghai*	*Manila*	*Bangkok*	*Krakow*	*Santiago*
Population	12,000,000	13,452,000	8,900,000	5,894,000	825,000	5,236,000
Exposed population (percent)	80	70	80	80	80	80
Mortality rate per 1,000 population	10	7	7	7	10	6
Estimated change in exposure ($\mu g/m^3$)						
Inhalable particulates (PM_{10})	27	72	35	30	38	53
Sulfur dioxide (SO_2)	0	17	0	0	15	0
Health effects (cases)						
Premature deaths	2,189	3,979	1,466	822	211	1,054
Respiratory hospital admissions	3,127	8,121	2,993	1,677	301	2,642
Asthma attacks	846,700	2,199,117	810,345	454,094	81,497	715,448
Emergency room visits	61,337	159,308	58,703	32,895	5,904	51,828
Restricted activity days	10,937,138	28,406,817	10,467,525	5,761,239	1,052,733	9,241,705
Lower respiratory illness in children	118,895	308,804	113,790	66,835	11,444	100,464
Respiratory symptoms	34,808,630	90,407,782	33,314,037	18,335,769	3,350,437	29,412,732
Chronic bronchitis	7,973	20,709	7,631	4,276	767	6,737
Cough days	0	764	0	0	48	0
Chest discomfort days	0	1,141,079	0	0	72,270	0

Source: Authors' calculations.

impacts is assumed. This, however, does not make any noticeable difference in the valuation of social costs; the direct impact on health of SO_2 is very small in comparison with the effect of sulfates, which is reflected through PM_{10} exposure.

4 Valuation of Health Effects

Valuation of health effects is a critical component in assessing the social costs of pollution: it allows the performance of cost-benefit analysis of pollution control measures and provides a basis for setting priorities for actions. This chapter reviews valuation approaches for premature death and illness caused by air pollution.

Mortality

Valuation of a statistical life

The effects of air pollution on mortality can be assessed by using the *value of a statistical life* (VOSL). It is important to be clear as to what the VOSL does and does not attempt to measure because the notion of valuing human life is so controversial in public discussion. The VOSL does not purport to measure the compensation that would be required by or should be paid to an average person who dies in a road accident or a plane crash. Nor does it imply how much a fatally ill average person would agree to pay for a miracle of recovery, given such a choice. It is derived by considering a different problem.

As an example, suppose that a government is trying to assess what kind of building standards should be satisfied in an area that is prone to intermittent but usually moderate earthquakes and that has a total population of 10 million people exposed to earthquakes. It is estimated that a particular set of standards will, on average, halve the number of deaths from earthquake damage, from 2 deaths to 1 death per million people per year. The annualized cost of meeting the stricter standards amounts to US$20 million per year, after allowing for the net saving on reconstruction in the aftermath of earthquakes.

The question then is whether the government (or the population) considers that the expenditure of US$20 million per year is justified in relation to a reduction in the risk of mortality equivalent to the loss of 10 statistical lives per year averaged over a period of 10 to 20 years. No one can be certain when or even whether the reduction in lives lost to earthquakes will occur or who will turn out to be the beneficiaries of the stricter building standards. That is why we refer to the value of a statistical life: the focus is on *willingness to pay* (WTP) to reduce certain kinds of risk to which a particular population is exposed. If willingness to pay exceeds a value of US$2 million per statistical life, the benefits of imposing the stricter building standards may be judged to exceed the costs that will be incurred. A much lower value per statistical life would imply that the benefits of stricter building standards do not exceed the costs involved, unless there are other considerations that have not been taken into account.

Everyday individual actions in which people trade money for a small reduction in personal safety can be used to infer the value of a statistical life. A variety of valuation techniques has been used to estimate this

value: labor-market (hedonic) studies, the contingent valuation method (CVM), and various types of market-based analysis. Labor-market studies generally attempt to infer the compensation required in exchange for the increased risks associated with particular occupations while standardizing for all other attributes of the job and the worker. The contingent valuation method asks individuals hypothetical questions related to their willingness to pay for reductions in their risk of encountering particular hazards. The market-based approach attempts to infer willingness to pay for reductions in risk from the purchase of goods whose only purpose is to reduce the risks confronting an individual.

The literature on the VOSL, or on willingness to pay to avoid a statistical premature death, is relatively well developed, and several analyses have reviewed the empirical estimates, which are mainly from the United States (see, for example, Fisher, Chestnut, and Violette 1989; Miller 1990; Viscusi 1992, 1993; TER 1995). The two most complete surveys of the existing literature suggest a mean VOSL (in 1990 dollars) of US$3.6 million (IEI 1992) to US$4.8 million (USEPA 1997). Note that these estimates are based not on an average for the entire literature but on a mean of the "best fit" distribution for a very carefully screened group of 26 studies from which flawed analyses have been excluded. Five of the 26 studies are CVM studies; the rest are labor-market (wage-differential) studies.

There is also a substantial literature on the valuation of life that relies on the *human capital approach*. Human capital is the present value of future labor income. The human capital and WTP approaches are not entirely unconnected. In particular, theory shows that human capital provides a lower bound to WTP (see, for example, Cropper and Sussman 1990). However, the "consumer surplus" from living can be shown to exceed human capital by many times. (Compare the mortality cost estimates using the human capital approach in Table 4.1 with the WTP estimates of US$3.6 million–$4.8 million.) Although seemingly straightforward, the application of the human capital approach to developing countries can be problematic because of distorted wages, cross-subsidization of public services, difficulties in valuing various homemaking services, high unemployment rates, and so on. Given the wide disparity between the two measures, it is preferable to concentrate on the task of transferring the WTP estimates into the context of lives lost through poor air quality in countries with different income levels.

This paper attempts to stay on the conservative side within a range of reasonable estimates by using the lower value of US$3.6 million for the U.S. willingness to pay to avoid a statistical premature death. This value, however, can and should be used only as the basis for initiating the benefit-transfer process, which involves a series of adjustments that are described below.

Several uncertainties complicate the transfer of available WTP estimates into the context of premature deaths caused by air pollution in developing countries. One problem is that the existing results refer almost exclusively to lives lost as a result of accidents at work rather than from air pollution. It is argued that those who die in occupational accidents would have had many more remaining life years than those who die as a result of poor air quality and that those who are most at risk are already suffering from some underlying condition that may affect the values to be attached to their lives. It is also argued that the contextual effects are important and that the issue of latency should be considered. Finally (and most important in quantitative terms), the great difference in income levels between the surveyed U.S. populations and the "target" populations of the developing countries requires a significant adjustment in the U.S.-based VOSL. Since the assumed VOSL

determines the damage cost estimates that emerge from air pollution studies, these issues should be carefully interpreted in the approaches adopted for placing a monetary value on the health outcomes of exposure to air pollution.

Table 4.1 Human capital and mortality cost, by age, United States

Age group (years)	*Number of life years lost*	*Mortality cost (1992 U.S. dollars)*
Under 5	75	502,421
5–14	68	671,889
15–24	57	873,096
25–44	42	785,580
45–64	25	278,350
65 and older	10	22,977
All ages	12	143,530

Note: Cost estimates are based on life expectancy at the time of death and include labor force participation rates, average earnings, the value of home-making services, and a 6 percent discount rate used to convert figures into their present-value counterparts.
Source: U.S. Institute for Health and Aging.

Age effects, underlying health conditions, and the VOSL

If age effects are important in determining the VOSL, and if the age profile of respondents to VOSL questionnaires does not match the age profile of those at risk from poor air quality, the application of these VOSL estimates to the air pollution context will introduce a bias. Labor-market studies, from which VOSL estimates are usually drawn, measure compensation for risk of instantaneous death for people about 40 years old and thus value approximately 35 years of life (Viscusi 1993). A study of Philadelphia found that excess mortality attributable to air pollution falls almost entirely on the 65 and older age group (Schwartz and Dockery 1992a). Other studies that utilize age-specific mortality (except for Cropper et al. 1997) indicate that the great majority of deaths related to higher concentrations of particulates occur in the over-65 age category (Fairley 1990; Ostro et al. 1995; Saldiva et al. 1995; Sunyer et al. 1996). Because death from air pollution reduces life years by less than 35 years on average, the question is how a difference in the age distributions of those involved in WTP studies and those primarily affected by pollution would change the respective estimates of the VOSL.

There are two possible approaches for adjusting the VOSL to better reflect the preferences of those at most risk from air pollution. The first is outlined by Moore and Viscusi (1988), who present a study of risk in the context of the labor market in which one of the explanatory variables is not the risk of death but the expected loss of discounted life years. The discount factor is estimated within the context of the hedonic wage regression and is found to be in the region of 10 to 12 percent per year. Comparison, for example, of the remaining years of life for the average respondent in labor market studies and the average person in the over-65 age group in the United States (35 and 10 years lost, respectively) at a 10 percent discount rate gives an adjustment factor of 0.64.

The other possibility is to use the results of studies in which the responses to WTP questions are categorized by age group. The adjustment factor would be the ratio of the VOSL estimates for the over-65 group to the VOSL estimates for all replies. Jones-Lee, Hammerton, and Philips (1985) found this factor to be 0.75. The ratios, however, are at least as uncertain as the VOSL estimates, and the range of potential ratios is very large.

A particular benefit of the first approach, for the purposes of our analysis, is that it addresses a concern regarding the uncertainty of transferring the results of dose-response studies into a different context, as highlighted by the Cropper et al. (1997) study of air pollution in Delhi. That study found that although mortality risk due to exposure to particulates (measured as total suspended particulates, TSP) in Delhi is considerably lower than in the United States, the number of life years lost is similar. This result is not merely coincidental: the loss of a greater number of life years per average death from air pollution occurs precisely because of the same age distribution of deaths and major mortality causes that may account for a lower air pollution–related mortality risk for the entire population. Thus, the use of the central estimate from PM_{10}-based mortality studies, as suggested in the previous section, in combination with adjustment of the VOSL for number of life years lost, will result in a more robust assessment of the mortality costs in cases like that of Delhi.

Disability-adjusted life years (DALYs)

This volume further advocates alignment of the economic approaches to valuing sickness and premature death with the concept of *disability-adjusted life years* (DALYs), described in Box 4.1. The VOSL obtained from labor-market studies can be combined with the corresponding number of DALYs lost in order to estimate the implicit value per DALY. The respective VOSL is then adjusted by use of an average number of DALYs lost in air pollution studies (as well as in any other specific study).

According to the age distribution of DALYs, the VOSL from U.S. labor market studies that represent people about age 40 corresponds to 22 DALYs lost, while an average death at age 65 (assumed to be a mean age of those fatally affected by particulates) corresponds approximately to 10 DALYs lost. This implies that the value per DALY in the United States is US$164,000 and that willingness to pay to avoid a premature death due to air pollution should be scaled down to 45 percent (10/22) of the mean VOSL, or a value of US$1.6 million. This is a far greater adjustment than the 64 percent based on a simple discounting of life years lost at a rate of 10 percent. The reasons for the difference are (a) use of a much lower discount rate when calculating DALYs; (b) different social values assigned to a year of life at different ages; and (c) different weights given to healthy years and years lived with disability, which at older ages account for an increased proportion of total years lost due to premature death. The incorporation of the last factor into the DALY measure is important because it addresses another issue in the debate about the relationship between the mean VOSL and the value of an average death caused by air pollution: the willingness to pay of the chronically sick.

It is widely believed that those who succumb to the effects of poor air quality are likely to be suffering from some underlying health condition and that some number of acute deaths from exposure to particulates merely represents the "harvesting effect." From the perspective of our approach to adjusting the mean VOSL, the issue of underlying health conditions translates into the question of whether people who die from air pollution causes have more severe disabilities (across all health states) than other people from the same age group (65 and older for rich countries) and thus whether the number of DALYs lost associated with such a death would be smaller than for an average death from this age group. Unfortunately, there is no information on which to base a definite answer, but the difference is unlikely to be nearly as substantial as for the mean VOSL.

Box 4.1
Measuring the Burden of Disease: The Concept of DALYs

DALYs (disability-adjusted life years) are a standard measure of the burden of disease. The concept combines life years lost due to premature death and fractions of years of healthy life lost as a result of illness or disability. A weighting function that incorporates discounting is used for years of life lost at each age to reflect the different social weights usually placed on illness and premature mortality at different ages. The combination of discounting and age weights produces the pattern of DALYs lost by a death at each age. For example, the death of a baby girl represents a loss of 32.5 DALYs, and a female death at age 60 represents 12 lost DALYs. (Values are slightly lower for males.)

Several reservations about the use of DALYs are commonly cited. These should be clearly understood when interpreting the estimates.

- Attention to differences among morbidity states is limited. DALY calculations do take into account the duration, incidence, and prevalence of different gross stages of morbidity, following a scale of "severity" within a disease. However, social, cultural, and economic contexts are not considered for the different disabilities, and calculation problems for conditions with states of remission and relapse (such as cancer, malaria, and hypertension) have not yet been resolved.
- DALYs apply age weighting, a practice that implies value differences within a community depending on the age of onset of the disability or fatality. Alternative valuations have proposed (a) expenditure-sensitive age-weighting, (b) weighting of the two age extremes that require the greatest caregiving, and (c) differential weighting functions by gender or urban-rural residence.
- Of the nonhealth characteristics of the individual affected by a health outcome, only age and sex are considered. DALYs are blind to predisposing features that are biological (genetics), behavioral (smoking, drinking), cultural (ethnicity, caretaking of elderly), or economic (access, ability to purchase pharmaceuticals, training of medical personnel).
- Classification of mortality by the underlying cause of death fails to consider either the compounding disability status of comorbidity or the synergistic relationships among diseases. DALYs do not take into account the contribution of one disease to an increase in risk for a second disease or a group of other diseases.

Despite these limitations, the use of DALYs as a measure of the burden of disease provides a consistent basis for systematic comparison of total disease burdens across various populations and a tool for analyzing the cost-effectiveness of alternative interventions designed to improve health and generate large improvements in the health status of poor households in the developing world.

Source: Murray and Lopez (1996); see also World Bank (1993); Anand and Hanson (1997).

Contextual effects, latency effects, and the valuation of changes in life expectancy

It is generally accepted that the value placed by individuals on the avoidance of risk depends on the nature of the risk. Current VOSL estimates do not account satisfactorily for the characteristics of different risks. Moreover, most, if not all, estimates are calculated in the context of job- or transport-related risks, and these contextual differences should certainly be considered when trying to apply existing VOSL estimates to environmental policy analyses. One major difference between risks posed by air pollution and risks posed by traffic or occupational accidents is that the former are involuntary. Increases in controllable risks are likely to prompt greater avertive activity, reducing the exposure of the individual to the point at which the additional costs of the avertive behavior equal the expected benefits at the margin. This explains why an increase in controllable risks may be valued less than an increase in uncontrollable risks. The extent to which this difference undervalues the cost of air pollution is uncertain.

Another important characteristic of air pollution is that it often presents latent rather than immediate risks. Cropper and Sussman (1990) convincingly demonstrate that willingness to pay for a reduction in future risks is to be discounted at the consumption rate of interest. An additional complexity is that in dealing with latent risks, it may be difficult to separate issues relating to the quantity of life from those relating to the quality of life. Individuals may experience several years of pain before they die. Considering the pain and suffering of a prolonged terminal illness, one might expect that the willingness to pay to reduce these risks would be rather greater than for reducing risk of death following an automobile accident.

This issue of latency has particular importance for air pollution studies in light of the findings reviewed in the previous chapter showing that most premature deaths from particulate concentrations are from chronic rather than acute disorders. The prevalence of latent effects of exposure to particulates over acute effects, as revealed by chronic exposure studies, along with the controversy about valuing "harvested" deaths in the acute exposure studies, has led to a search for another approach to measuring the effect of air pollution on human health and mortality risk. Such an approach aims at quantifying and valuing the changes in life expectancy of the exposed population caused by variations in air quality. It deals with both chronic effects and the "harvesting" effect by comparing the average life expectancies of individuals exposed to different concentrations of particulates over a long period.

The *life expectancy approach* involves (a) estimating the change in life expectancy by age group implied by the change in ambient particulates; (b) establishing willingness to pay for the change in life expectancy by age group; and (c) multiplying these two values by each other and by the population in each age group and adding the results (ASEP 1997). Generally, this approach to valuing changes in life expectancy attributable to long-term exposure to air pollution seems very promising. First, it addresses the uncertainties of adjusting, for the air pollution context, willingness to pay to avoid contemporaneous risks at the prime age. Second, it may be politically more acceptable to explicitly incorporate the value of a change in average life expectancy into the design of environmental policies than to use the politically sensitive VOSL.

The main problem here is the lack of empirical evidence regarding willingness to pay for an increase in life expectancy. All but a few mortality valuation studies assess accidental death risks rather than latent risks that may cause a premature death many years from the present. The first study that valued changes in life expectancy—a contingent valuation survey in Japan conducted by Johannesson and Johansson (1996)—contains a large number of uncertainties that call for further research. A recent CVM study by Krupnick et al. (1999), also for Japan, proposes an approach to valuing mortality risks from air pollution that involves contemporary risks for older people and long-term risks for younger people. Whereas older people (over age 70) face an immediate risk of death from exposure (and an immediate reduction of risk from a reduction in exposure), the WTP of younger persons should reflect what a person would pay *today* for a *future* risk reduction. The study develops and tests a valuation survey that recognizes these two different types of risk. It reports a much lower implied VOSL than that in most labor-market analyses: US$551,000–$1,262,000 for people under age 70, and US$288,000 for people over age 70.

It is important to stress, however, that the results of valuing long-term latent effects and life expectancy are supposed to be used in

combination with dose-response studies of chronic exposure. Thus, a possible decrease in valuation parameters will be counterbalanced by a sharp increase in the magnitude of health impacts linked to air pollution (see Chapter 6).

The balance of evidence therefore allows us to infer that the use of an adjusted VOSL of US$1.6 million in combination with a change in mortality risk of 0.84 per 10 $\mu g/m^3$ change in PM_{10} concentration, as implied by acute exposure studies, yields results that are prudent and are unlikely to overstate the "true" social costs of mortality associated with exposure to ambient particulates. This inference is based on analysis of all major uncertainties involved in the valuation of a premature death from air pollution causes, as well as on a large body of other studies and approaches dealing with these issues.

Morbidity

Air pollution also affects human morbidity, and the valuation of illness and disability is important for assessing the full social costs of air pollution. The literature on willingness to pay to avoid morbidity effects is limited in scope and is based entirely on U.S. data. An alternative that is often employed for valuing morbidity is the cost of illness (COI) approach, which uses estimates of the economic costs of health care and lost output up to recovery or death. These comprise the sum of direct costs (hospital treatment, medical care, drugs, and so on) and indirect costs—the value of output lost, usually calculated as the wage rate multiplied by lost hours and often using an imputed wage for home services (see Cropper 1981). Although the COI approach is often viewed as easily applicable to any country, subsidized or inadequate medical services and drug supplies in many developing countries make it difficult to calculate the economic costs of health care. More important, COI fails to account for the disutility of illness, which is likely to be a major component of willingness to pay for reducing the risk of falling sick. As a result, most of the existing work on valuing the health effects of air pollution uses a combination of the WTP approach (where estimates are available) and the COI approach (where WTP estimates are lacking).

The approach taken in this paper is consistently to use WTP estimates to value a variety of morbidity outcomes. As this analysis points out, however, some of the costs of illness may not show up in WTP estimates, either. Costs of health care borne by the public sector, for example, will not be reflected in individual willingness to pay. Therefore, in future work on morbidity costs, some components of COI estimates may be used to supplement WTP estimates, to reflect the full costs to society. When reconciling COI and WTP estimates, great care must be taken to avoid double-counting (see the discussion below).

Valuation of chronic bronchitis

Chronic bronchitis is the only morbidity end-point quantified in the previous chapter that may last from the beginning of the illness throughout the rest of the individual's life. The valuation of this illness should therefore be carried out separately from that of the other health effects related to air pollution. Two studies provide estimates of willingness to pay to avoid chronic bronchitis, using the CVM analysis: Viscusi, Magat, and Huber (1991) and Krupnick and Cropper (1992). In establishing the "best" estimate from these two studies, we followed the approach adopted in the recent USEPA review of the costs and benefits of cleaner air (USEPA 1997). The Viscusi, Magat, and Huber study uses a larger and more representative sample of the general population. However, it defines a case of chronic bronchitis that is much more severe than an "average" case from the Abbey et al. (1993) study used to establish the dose-

response relationship (see Chapter 3). Hence, the USEPA report starts with willingness to pay to avoid a severe case of chronic bronchitis, as described by Viscusi, Magat, and Huber (1991), and adjusts it downward to reflect a less severe case of pollution-related chronic bronchitis and the elasticity of willingness to pay with respect to severity. The latter is derived from the Krupnick and Cropper (1992) study, which estimated the relationship between willingness to pay and the severity level. This approach resulted in a mean willingness to pay of US$260,000 (in 1990 dollars), which is regarded as a reasonable value in relation to the COI estimates for chronic bronchitis reported by Cropper and Krupnick (1990). Specifically, the WTP estimate of US$260,000 is from 3.4 to 6.3 times the full COI estimates, depending on age (30 to 60 years).[15]

It is important to keep a consistent ratio between the VOSL and willingness to pay to avoid a chronic illness. Since USEPA (1997) uses a VOSL of US$4.8 million, whereas this study adopts a lower estimate of US$3.6 million, we downsized the willingness to pay to avoid a new case of chronic bronchitis accordingly and used the base value (before adjustment for income) of US$195,000 in our calculations.

Valuation of acute morbidity effects

Proceeding initially on the assumption that all the costs associated with morbidity effects are privately borne, one solution for dealing with the paucity of WTP literature and the inadequacy of COI literature is to integrate the health status index literature with the available WTP literature. The health status index literature attempts to measure individuals' perceptions of the quality of well-being (QWB) on a scale ranging from 0 (death) to 1 (perfect health). Any health state can be evaluated by considering its impact on various symptoms; its effect on social activity, physical activity, and mobility; and its duration. By these means, the conceptually appropriate WTP values can be obtained for each acute morbidity impact that has been described in the health status index literature and investigated in the air pollution literature, given the established correlation between WTP values and QWB scores. This approach, proposed by TER (1996), is followed and extended here. In making such extrapolations, it is important to distinguish between acute and chronic effects because the very fact of irreversibility of a poor health state adds a significant component that will not be captured by estimates of willingness to pay to avoid temporary acute disorders.

Once the relationship between the health status index and willingness to pay is found, willingness to pay for any condition that can be described using the QWB score can be predicted—even those conditions for which no valuation experiments are available. This is a potentially very useful application, since many health states for which no WTP values are available have been investigated in the epidemiological literature. The procedure for estimating the predicted willingness to pay for avoiding morbidity end-points identified in the air pollution literature is described in Annex C.

Table 4.2 compares predicted WTP with the WTP estimates encountered in the epidemiological literature. The table also includes two COI-based measures for respiratory hospital admissions (RHA) and emergency room visits (ERV).[16] The divergence between the predicted and published WTP values is small in most cases. However, the COI exceeds predicted WTP for ERV and RHA (although the COI values do fall within their respective confidence intervals). This finding may well reflect the existence of large publicly borne costs associated with hospital treatment or mandatory sick pay.

The private and the social costs of illness

The use of individual willingness to pay to avoid particular health states (as advocated in the preceding sections) becomes problematic when it is recognized that some of the costs of health care are borne by the public sector. These costs are not reflected in individual willingness to pay. No rational individual would be willing to incur expenditure to avoid falling sick simply because of the costs to the state or the person's employer. The social costs of illness comprise the private willingness to pay plus the publicly borne costs. The COI, by contrast, includes treatment cost plus loss of earnings, although not the disutility from illness. It is clear, therefore, that some components of COI estimates may be used to supplement WTP estimates and so reflect the full costs to society. A similar problem occurs when private insurance is available for the costs of health care and loss of earnings in the event of prolonged illness. An individual who has already purchased insurance will discount loss of earnings and health care costs in determining individual willingness to pay to avoid the health impact. These costs are paid by the insurance company or are shared across a larger group of people. Both problems are significant; for example, in the United States 68 percent of all health-related expenditures is paid by third parties (Chestnut and Violette 1984).

The ideal solution to the problem of the publicly borne costs of ill health would be to decompose individual WTP into its constituent parts (disutility of illness, loss of earnings, and treatment costs). The disutility of illness element could be explained by reference to symptoms and then valued by means of surveyed or predicted WTP. The remaining elements (treatment costs and loss of earnings) could be estimated separately and added to the pure utility costs. The currently available evidence, however, makes no attempt to decompose WTP into different motivations. Mechanically adding COI and WTP may result in substantial double-counting. In this study we simply use the predicted WTP estimates that correspond to the privately borne costs of morbidity effects and can be viewed as a lower bound to the full social costs.

Income Effects

A major uncertainty that complicates the application to developing countries of WTP estimates for industrial market economies arises from differences in income levels. One of the fundamental issues in valuing the reductions in risk is that willingness to pay rises with income. Since the existing VOSL estimates are taken almost exclusively from the United States, there is a clear need to adjust the VOSL for income effects before applying the results to developing countries.

Table 4.2 Comparison of economic values for morbidity effects

Morbidity effect	*Study*	*Duration (days)*	*Study type*	*Value (1990 U.S. dollars)*	*Predicted WTP (1990 U.S. dollars)*
Respiratory hospital admission	Cropper and Krupnick (1990)	9.5	COI	6,589	4,302
Emergency room visit	Rowe et al. (1986)	1	COI	220	131
Bad asthma day	Rowe and Chestnut (1985)	9.5	WTP	525	324
Cough day	Tolley et al. (1986)	1	WTP	32	44
Eye irritation	Tolley et al. (1986)	1	WTP	35	42

Source: Compiled by D. Maddison.

The most general way of adjusting for differences in income levels is to calculate country-specific valuations for country j using the formula:

$$\log(V_k) = r \log(Y_k/Y_{US}) + \log(V_{US}) \quad (4.1)$$

where Y_k and Y_{US} are the per capita incomes of country k and the United States; V_k and V_{US} are valuation parameters for health end-points in country k and the United States; and r is the income elasticity of the relevant willingness to pay.

The literature on the income elasticity of willingness to pay for reducing the risk of damage to health is, however, extremely sparse. There appears to be only a few empirical analyses in the literature that investigate the income elasticity of willingness to pay for reductions in risk. Loehman and De (1982) estimate an income elasticity in a range between 0.26 and 0.6 in their study of willingness to pay to avoid respiratory symptoms associated with air pollution in Tampa, Florida. Jones-Lee, Hammerton, and Philips (1985) point to a rather low elasticity of around 0.4. Biddle and Zarkin (1988) infer an income elasticity of willingness to pay of 0.7, whereas Viscusi and Evans (1990) suggest a much higher income elasticity of 1.1. Because the standard errors associated with these estimates are not available, a simple average of the studies yields an income elasticity of 0.65. A recent study of the relation between income and the VOSL from wage-differential studies in industrial countries found a mean elasticity of 0.55 (Day 1999). By comparison, cross-sectional analysis of per capita expenditures in the 1980 International Comparisons Project found an income elasticity of demand for medical goods and services of 1.05.

It is important to note the acute sensitivity of the social costs of ill health to the value of this parameter. Using an elasticity of 0.4 or 1.1 makes a difference of nearly 20 times in the income adjustment for India. Furthermore, the VOSL for the United States that we have used is equivalent to about 160 times the average per capita U.S. gross national product (GNP) in 1990. Applying equation (4.1) by using average per capita GNP for Y_k and the elasticity of 0.7 gives ratios for (VOSL/Y_k) of over 300 for Poland, over 400 for the Philippines, and over 500 for India in 1993. The implication that the relative willingness to pay for reductions in the risk of mortality in India is more than three times the U.S. level seems an extremely strong and probably counterintuitive assumption.

It is more reasonable to infer that estimates of the income elasticity of willingness to pay to avoid risks of death or ill health are not robust to large income differentials, on the grounds that these estimates refer to cross-sectional differences in income within a country or a group of similar countries and should not be applied to vast differences across countries. To maintain a degree of conservatism in this valuation exercise, we have thus chosen to assume a higher income elasticity of 1 for both the VOSL and morbidity cost estimates, so that attention is focused purely on differences in income.

There is also an issue of whether income in developing countries should be measured in U.S. dollars at a market exchange rate or with the use of a purchasing power parity (PPP) conversion rate when transferring the VOSL estimates from industrial countries. International comparison of a variety of development indicators for a large number of countries found, for example, that PPP-based estimates of per capita gross domestic product (GDP) provide much better explanations of variations in key health indicators, such as life expectancy and infant mortality, than income measured at a market exchange rate (see Summers and Heston 1995). PPP-based estimates of GDP are, however, often criticized

as not very reliable for developing countries. In this study, all incomes are converted to U.S. dollars using the respective market exchange rates. It is worth noting that the use of PPP-based estimates, which is advocated in a number of studies (for example, Markandya 1991; ASEP 1997), would considerably increase the social costs of mortality and morbidity, especially in low-income countries such as India or China (see Chapter 6).

Additional evidence that the use of an elasticity of 1 in combination with market exchange rates for converting incomes into U.S. dollars is likely to yield very conservative results for developing countries comes from recent studies on willingness to pay to avoid respiratory illness in Bangkok (Chestnut, Ostro, and Vichit-Vadakan 1997) and Taiwan, China (Alberini et al. 1997; Alberini and Krupnick 1998). The results suggest that *as a share of income*, willingness to pay to prevent a respiratory illness tends to be higher in both studies than similar estimates in the United States, and thus the income elasticity of willingness to pay to avoid illness is well below 1.

Finally, a recent wage differential study for India found a VOSL ranging from US$150,000 to US$360,000 (Simon et al. 1999). This is a much higher value than the US$50,000 that can be derived by scaling down the U.S. VOSL by the difference in incomes (GDP per capita), measured at a market exchange rate, although it is quite consistent with the use of an elasticity of 0.65. The use of the PPP exchange rate for converting Indian GDP per capita to U.S. dollars yields a compatible estimate of US$360,000 for the VOSL in India. The latter finding, together with other evidence, suggests that the use of a PPP exchange rate with an elasticity of 1 or slightly higher may be a better approach.

International Comparisons of Health Costs and DALYs

High income elasticity and great income disparities complicate cross-country analysis of the health costs of air pollution when these are expressed in monetary terms. For example, air pollution from fuel burning causes three times more premature deaths in Shanghai than in Santiago, but the monetary damage is larger for Santiago, which has the highest income level in the sample. One way of comparing the severity of air pollution across cities and countries is to present the health costs as a share of respective incomes (average city income or country GDP per capita). Another approach is to use the concept of disability-adjusted life years (DALYs), explained in Box 4.1, above.

An aggregate measure such as DALYs is able to reduce all health effects—mortality and various morbidity end-points—to one denominator. In this it is similar to the economic valuation procedures, but it is independent of income. Expression of the health burden of air pollution in DALYs has also the advantage of direct comparison with the overall burden of disease in developing countries, as well as with diseases from other major environmental causes (e.g., water-related diseases). This is possible because of the significant amount of work by public health specialists on generating DALY estimates for various countries.

In this exercise we have attempted to express the health burden of fuel combustion in DALYs. For mortality due to air pollution causes, the approach is straightforward: use of 10 DALYs lost per death, a value that corresponds to an average death at the age of 65 in the United States and was used to scale down the respective VOSL. Converting air-pollution-induced morbidity to DALYs is a tougher challenge because of the lack of

literature relating morbidity end-points assessed in air pollution dose-response studies to lost DALYs.

An important message from this review of valuation approaches is an evolving convergence between the methods for assessing the burden of ill health being devised by economists and by public health specialists. This is evident from the attempt to combine the measure of DALYs with the age- and context-specific VOSL and the integration of willingness to pay to avoid illness with the health status index. This integrative tendency should receive further support, as it promotes greater acceptance of the aggregate measures of the burden of disease, provides for consistent assessment of environmental health priorities, and unites public efforts to reduce the risk of exposure to environmental hazards.

The close link between health status in the public health literature and economic valuation that was consistently adopted in this exercise enabled us to assign DALYs to morbidity outcomes caused by air pollution. The approach used was to adjust the number of DALYs assigned to the mortality outcome proportionally to the ratio between the value of an air pollution–related death and valuation parameters for morbidity effects.

Summary of Valuation Parameters and Results for the Six Cities

Table 4.3 contains the base valuation parameters for all health states adopted in this study for U.S. 1990 GNP per capita (US$21,790). Table 4.4 shows the social costs of these health outcomes and the values per case in U.S. dollars for each of the six cities, after adjustment for income differences with the United States. Table 4.5 converts these health outcomes into loss of DALYs. Note the difference in the ranking of the cities by the monetary values of the health costs and by the health burden expressed in DALYs.

Table 4.3 Base values for health effects used in the study (for the U.S. income level, 1990)

Health status	*DALYs lost per 10, 000 cases*	*WTP-based monetary value per case (1990 U.S. dollars)*
Premature death	100,000	1,620,000
Chronic bronchitis	12037	195,000
Respiratory hospital admission	264	4,225
Asthma attack	4	63
Emergency room visit	3	126
Restricted activity day	3	53
Lower respiratory illness in children	3	44
Respiratory symptoms	3	44
Cough day	3	44
Chest discomfort day	3	50

Source: Authors' calculations.

Table 4.4 Social costs of health damages from fuel use: Six cities

	Mumbai	*Shanghai*	*Manila*	*Bangkok*	*Krakow*	*Santiago*	*All six cities*	*Percent*
Population	12,000,000	13,452,000	8,900,000	5,894,000	825,000	5,236,000	46,307,000	
Average income (U.S. dollars per capita)	450	980	1,425	3,225	2,712	3,487	1,528	
Unit values (U.S. dollars per case)								
Premature death	33,456	72,859	105,943	239,766	201,626	259,245		
Respiratory hospital admission	88	192	280	633	532	684		
Asthma attack	1	3	4	9	8	10		
Emergency room visit	3	6	8	19	16	20		
Restricted activity day	1	2	3	8	7	8		
Lower respiratory illness in children	1	2	3	7	5	7		
Respiratory symptoms	1	2	3	7	5	7		
Chronic bronchitis	4,027	8,770	12,752	28,861	24,270	31,205		
Cough day	1	2	3	7	5	7		
Chest discomfort day	1	2	3	7	6	8		
Health costs (thousands of U.S. dollars)								
Premature deaths	73,226	289,930	155,347	197,012	42,477	273,291	1,031,283	39
Respiratory hospital admissions	276	1,561	837	1,061	160	1,807	5,703	0
Asthma attacks	1,102	6,231	3,339	4,234	639	7,213	22,757	1
Emergency room visits	160	903	484	613	93	1,045	3,297	0
Restricted activity days	11,971	67,712	36,281	45,192	6,944	78,383	245,484	9
Lower respiratory illness in children	108	611	327	435	63	707	2,252	0
Respiratory symptoms	31,630	178,907	95,860	119,405	18,348	207,101	651,251	25
Chronic bronchitis	32,109	181,617	97,312	123,412	18,626	210,238	663,314	25
Cough days	0	2	0	0	0	0	2	0
Chest discomfort days	0	2,540	0	0	445	0	2,986	0
Total health costs	*150,580*	*730,014*	*389,787*	*491,366*	*87,795*	*779,787*	*2,629,329*	100

Source: Authors' calculations.

Table 4.5 Health burden of fuel use: Six cities

	Mumbai	*Shanghai*	*Manila*	*Bangkok*	*Krakow*	*Santiago*	*All six cities*	*Percent*
Population	12,000,000	13,452,000	8,900,000	5,894,000	825,000	5,236,000	46,307,000	
Average income (U.S. dollars per capita)	450	980	1,425	3,225	2,712	3,487	1,528	
Health burden (DALYs)								
Premature deaths	21,887	39,793	14,663	8,217	2,107	10,542	97,209	41
Respiratory hospital admissions	83	214	79	44	8	7	498	0
Asthma attacks	329	855	315	177	32	278	1,986	1
Emergency room visits	48	124	46	26	5	40	288	0
Restricted activity days	3,578	9,294	3,425	1,885	344	3,024	21,549	9
Lower respiratory illness in children	32	84	31	18	3	27	196	0
Respiratory symptoms	9,454	24,555	9,048	4,980	910	7,989	56,936	24
Chronic bronchitis	9,597	24,927	9,185	5,147	924	8,110	57,891	24
Cough days	0	0	0	0	0	0	0	0
Chest discomfort days	0	349	0	0	22	0	371	0
Total DALYS	*45,009*	*100,195*	*36,792*	*20,494*	*4,354*	*30,079*	*236,923*	*100*

Source: Authors' calculations.

5 Valuation of Nonhealth and Climate Change Effects

This chapter summarizes evidence from a diverse body of studies that have attempted to value local nonhealth, regional, and global climate effects.

Local Nonhealth Effects

Apart from its impact on health, air pollution is blamed for damaging buildings, soiling clothes and historic monuments, and reducing visibility. These costs are thought to be small in relation to the health-related costs. The main difficulties in valuing them are lack of knowledge concerning the relevant income elasticities of willingness to pay (WTP) to avoid these effects and doubts about the methodologies underlying those studies that have been undertaken—whether they measure true WTP or some upper or lower bound on it.

Visibility

For visibility, the problem is to disentangle WTP motivated by the desire to improve visibility from WTP arising from other, more general, motives. The procedure is to meta-analyze the literature on willingness to pay for an increase in visibility range in such a way as to distinguish between studies that deal with the problem of "embedding" and those that do not. Embedding refers to the difficulty that people encounter when trying to isolate one motive from a group of possible motives underlying their WTP. This analysis indicates that a number of studies should be discounted for policy purposes because they fail to isolate visibility impacts from more general impacts such as health effects and soiling. Next, a simple relationship is utilized to link visibility range with the mass of particulates in the atmosphere. By linking this equation with the equation on willingness to pay for improvements in visibility range, it is possible to infer WTP for a unit reduction in particulates insofar as the effect of the reduction on visibility range is concerned. This results in a marginal damage function that, unusually, is downward sloping: people are willing to pay less to reduce particulate concentrations when visibility is already badly impaired. It is further assumed that visibility is a "luxury good." Both considerations suggest that WTP in highly polluted cities in developing countries will be low.

Soiling

A variety of methodologies has been employed to derive dollar values for the role of particulates in soiling. Some, such as methodologies based on the contingent valuation method and the household production function approach, attempt to measure WTP. Others, such as those based on observed cleaning frequencies, seek to estimate a lower bound: since the cost of achieving a given degree of cleanliness increases with particulate concentrations, the individual rationally "buys" less cleanliness. Hence the observed increase in cleaning expenditures is a lower bound on WTP. A problem to be considered is that the cost of cleaning includes own-labor, not just the cost of cleaning materials. Studies that attempt to estimate WTP on the basis of purchases of cleaning

commodities alone may yield lower estimates than those based on increases in cleaning frequency if the cleaning tasks are valued at commercial rates. The income elasticity of expenditure on cleaning materials is determined from international expenditure data.

Materials damage

For damage to buildings and structures, the approach taken is to observe the maintenance cycle for various building components and to associate this cycle with a critical degree of surface recession of the material. Dose-response functions are available for different materials to indicate the length of time before a component needs to be replaced. The effect on component lifetimes of changes in the level of air pollution, and hence on annual replacement costs, can then be calculated. This approach can be shown to yield a lower bound on materials damage: since a reduction in air pollution is likely to reduce the costs of achieving a given standard of maintenance, more maintenance services and a higher standard of repair, would be desired. This leads to gains over and above those suggested by the replacement cost approach. Damage costs per unit of air pollution are derived by taking average damage costs in industrial countries and dividing by average air pollution monitor readings. These expenditures are adjusted to account for differences in the "acceptable" degree of materials damage in developing countries and for differences in the stock of building materials at risk.

Table 5.1 contains the proposed quantifications for all these effects, which are further explained in Annex D. Table 1.3, in Chapter 1, shows, for each city, nonhealth damages to the local population from fuel use, together with the health damage and global climate costs. These nonhealth damages are much smaller than health costs (about one-sixth of the latter). This estimate is consistent with other comparisons of health and nonhealth damages or benefits (e.g., Burtaw et al. 1997; USEPA 1997). It is worth noting that the estimates of damages from reduced visibility, soiling, and corrosion are based on rather old studies that may be not quite comparable with the very recent studies of health impacts and climate change costs used in this analysis.

Table 5.1 Base values for local nonhealth effects used in the study

Pollutant and physical impact	*Ambient level ($\mu g/m^3$)*	*Monetary value per person per $\mu g/m^3$ (1990 U.S. dollars)*
Total suspended particulates		
Visibility	50	0.80
	100	0.50
	150	0.30
	200	0.20
	250	0.10
Soiling		0.50
Sulfur dioxide (SO_2)		
Corrosion		0.45
Nitrogen oxides (NO_x)		
Corrosion		0.20

Note: Values are for the U.S. income level in 1990. They are adjusted for individual cities according to the difference between the city's income per capita and 1990 U.S. GDP per capita, using the following elasticities: for visibility, 1; for soiling, 0.9; and for corrosion, 0.65. See Annex D for details. Values may be converted from TSP to PM_{10} by multiplying them by 1.8.
Source: Authors' calculations.

Transboundary and Ecosystem Effects

Dose-response functions exist for ozone, which is widely held to be the most important pollutant for crop damage. For damage to forests linked either to ozone damage or to wet or dry acidic deposition, dose-response functions exist but are simplistic. Knowledge concerning damage to aquatic life is similarly limited. As in the case of forests, the damage is

likely to involve not only commercial losses but also diminished recreational use and nonuse benefits. Acid rain damages buildings and historic monuments, but there is not enough information to permit the valuation of this damage at present.

A characteristic of much transboundary pollution is that the damage so caused depends intimately on the characteristics of the location in which the pollution is deposited. For these reasons, cost estimates for acid rain damage cannot credibly be transferred—certainly not from temperate to tropical or subtropical zones. The damages depend too much on species tolerance and soil characteristics, and considerably less is known about these conditions in the developing world than in industrial countries. Moreover, social and cultural differences between countries suggest that the transfer of values relating to damage inflicted on the stock of cultural heritage could be very difficult. Nonetheless, although rapid assessment of transboundary impacts using figures taken from rich countries located in temperate zones is somewhat lacking in credibility, the available evidence suggests that these impacts are likely to be small in relation to other impacts occurring within large, densely populated cities (see, for example, EC 1995; Burtaw et al. 1997).

Global Climate Change

Despite the effort that has gone into researching the effects of potential climate change, the possible impacts remain highly uncertain. Most estimates have been derived largely by enumerating the impacts of climate change on the United States, placing monetary values alongside these impacts, and aggregating up to the rest of the world on the basis of shares of gross domestic product (GDP), population, or length of coastline. This yields a "global damage function" in which losses to world GDP are held to be some function of an index of global climate change (typically, global average temperatures relative to preindustrial levels). Other work has attempted an aggregation of region-specific damages based on regional models of impacts on agriculture and sea level. The valuation of climate change impacts is reviewed in the 1995 report of the Intergovernmental Panel on Climate Change (Pearce et al. 1996), and several more estimates have been produced since then. This recent work suggests that once adaptation to climate change has occurred, the losses of marketed goods (timber, agricultural produce, and so on) may be changed into gains, at least in the United States (Mendelsohn and Neumann 1999). The USEPA estimates a range of damage for marketed goods of –2.0 to +1.2 percent of U.S. gross national product (GNP) by 2080. These estimates exclude nonmarket impacts and are based on different assumptions about climate sensitivity, rates of warming, and vulnerability. Similar work (Maddison 1997) suggests that the amenity value of climate change may be substantial in many northern countries.

Notwithstanding their current shortcomings, these global damage functions can be used to derive corresponding shadow price estimates of the marginal emissions of greenhouse gases (GHGs). This derivation involves use of integrated assessment (IA) models. IA modeling exercises combine knowledge from different disciplines to determine how emissions of GHGs are transformed into GHG concentrations in the atmosphere and how rapidly these bring about the changes in climate that cause damages. The results of several such IA models, in terms of the shadow price of carbon emissions that they predict, are given in Table 5.2. It is important to note that different shadow price estimates emerge from model experiments involving different policy contexts, quite independent of the structure and parameterization of the IA model. Shadow price estimates derived from cost-benefit analysis of climate change yield estimates of

current-value marginal damage evaluated along the socially optimal level of abatement (see, for example, Cline 1993). By contrast, the marginal damage estimates present the current value of marginal damage assessed on the assumption that no abatement is undertaken (see, for example, Fankhauser 1994, 1995). These estimates are naturally higher, since the stock of emissions accumulates faster in the latter models. Whichever model is appropriate depends on the policy context, but, as Table 5.2 shows, the difference in the results between these two approaches appears to be small.

Several factors explain the variability of the estimates in Table 5.2. First, the effect of different discount rates is shown by the values of ρ, the utility discount rate. This rate discounts future utility, and it is usual to add it to the rate for discounting future income (consumption). Controversy surrounds the value of ρ, since some authors regard utility discounting as illicit; that is, they set $\rho = 0$. The effect is easily seen by comparing Fankhauser's estimates with utility discount rates of 0 and 3 percent; the difference is a factor of 9 in the damage estimate. The discount rate partly

Table 5.2 Estimates of marginal damage costs from global warming or of marginal primary benefits from optimal control of warming (U.S. dollars per ton carbon; base year prices, 1990)

Study	*1991–2000*	*2001–10*	*2011–20*	*2021–30*
Nordhaus (1991)	7.3	7.3		
Nordhaus (1994)				
With ρ = 0.03, best guess	5.3	6.8	8.6	10.0
With ρ = 0.03, expected value	12.0	18.0	26.5	
Nordhaus (1998)	5.0	7.1	9.3	11.7
Fankhauser (1995)				
With ρ = 0, 0.005, 0.03	20.3	22.8	25.3	27.8
With ρ = 0	48.8	—		62.9
With ρ = 0.03	5.5	—		8.3
Cline (1993)				
With s = 0	5.8–124	7.6–154.0	9.8–186	11.8–221.0
Peck and Teisberg (1992)				
With ρ = 0.03	10.0–12.0	12.0–14.0	14.0–18.0	18.0–22.0
Maddison (1994)	5.9–6.1	8.1–8.4	11.1–11.6	14.7–15.2
Eyre et al. (1997)				
With s = 0	142.0	149.0		
With s = 1	73.0	72.0		
With s = 3	23.0	20.0		
With s = 5	9.0	7.0		
With s = 10	2.0	1.0		
Tol (1999)	11.0	13.0	15.0	18.0
Roughgarden and Schneider (1999): lower bound = Nordhaus; upper bound = Tol	5.0–11	6.0–13	8.0–16.0	10.0–21.0

Note: ρ = utility discount rate; s = the overall discount rate. Eyre et al. estimates are for 1995–2004 and 2005–2014; the estimates here exclude equity weighting. Roughgarden and Schneider's ranges derive from inserting the models of Fankhauser (1995), Cline (1992), Titus (1992) and Tol (1995) into Nordhaus's Dynamic Integrated Climate Economy (DICE) model framework. The upper end of the range should, strictly speaking, coincide with the marginal damage estimates in Tol (1999). Note also that the table shows the considerable sensitivity of estimates to the discount rates.
Source: Pearce (2000).

explains Cline's large range, but his estimates also reflect very high estimates of damage and a very long-term time horizon. The "central" estimates are surprisingly similar. For 1991–2000 the damage estimate is around US$7–$15 per ton carbon, but the Eyre et al. estimates are about twice that figure. Interestingly, the most recent studies, by Tol and Nordhaus, suggest damage estimates lower than those previously estimated, although Nordhaus's estimate is fairly constant, at US$6–$9, taking his preferred "best guess" (Nordhaus 1998; Nordhaus and Boyer forthcoming). Tol's 1999 paper is also probably the most careful recent estimate. Fankhauser's model has considerable attractions because of its use of the discount rate as a random variable. This is shown here in the row with $\rho = 0, 0.005, 0.03$, that is, with a probability distribution assumed for ρ taking values of 0, 0.5, and 3 percent. Roughgarden and Schneider (1999) review four studies of *total* damages: Titus (1992); Cline (1992); Fankhauser (1995); and Tol (1995). Making slight modifications to these damage estimates, they derive *damage functions* that are then used to produce optimal carbon taxes (marginal damage estimates), as shown in the final row of Table 5.2.

As pointed out above, Mendelsohn and Neumann (1999) suggest *net benefits* for impacts on the market sector in the United States, and Mendelsohn et al. (1996) suggest that this conclusion may hold true for the world as a whole. This is not the position taken here. Also ignored is the effect of equity weighting on the estimates. For this debate, see Azar and Sterner (1996); Azar (1999); Fankhauser, Tol, and Pearce (1997, 1999); Tol, Fankhauser, and Pearce (1996, 1999).

Overall, marginal damages of US$5–$22 per ton carbon (in 1990 dollars) for 1991–2000 and US$6–$25 per ton carbon after 2000 seem defensible as a central range. This volume chooses an estimate of US$20 per ton carbon for the year 1993 (equivalent to US$18.20 in 1990 dollars), which corresponds to the higher end of the proposed range. Because of significant uncertainty, that value is not intended to reflect an upper bound on the possible damage from GHG emissions, but it is far from being a conservative estimate. The resulting damage cost estimates per ton of fuel are presented in Table 5.3. These marginal damage costs per unit of fuel were applied directly to fuel use inventory data for the six cities.

Table 5.3 Climate change damage costs associated with different fuel products

Fuel	*Damage cost at US$20 per ton carbon*
Hard coal	13.80/t
Lignite	5.70/t
Briquettes	15.70/t
Coal gas	19.50/t
Fuel oil	16.20/t
Distillate oil	16.50/t
Diesel	16.50/t
Gasoline	16.50/t
Liquefied petroleum gas	18.10/t
Natural gas	10.10/m^3
Fuelwood	5.70/t

Sources: IEA and OECD (1991); Pearce et al. (1996).

6 Summary of Methodological Issues

Having followed a series of steps in Chapters 2 through 5 and established links between the key parameters for damage assessment, we can easily track these links back and quantify damages per unit of emissions or fuel use. The final results of this damage assessment exercise are reported in Chapter 1. This chapter provides complementary information not given above, discusses the applicability to other studies of the techniques, assumptions, and findings of the assessment, and outlines further research and development needs.

Shortcuts for Rapid Damage Assessment

The method described in the preceding chapters provides a quick guide for estimating the order of magnitude of damages from a change in exposure to local pollutants.

Table 6.1 shows the levels of various types of damages associated with a 1 microgram per cubic meter ($\mu g/m^3$) change in exposure to inhalable particles (PM_{10}), sulfur dioxide (SO_2), and nitrogen oxides (NO_x), normalized on a population of 1 million with an average per capita income of US$1,000 per person, using the income elasticity of 1 for health impacts and the following elasticities for nonhealth impacts: for visibility, 1; for soiling, 0.9; and for corrosion, 0.65 (see Annex D). Other assumptions used in the table are the crude mortality rate (8 per 1,000 population), for calculating mortality cost; shares in total population of children under 14 (27 percent) and of asthmatics (5 percent), for morbidity cost; and the annual mean level of total suspended particulates, TSP (150 $\mu g/m^3$), for visibility cost. It is also assumed that the PM_{10} mix is typical of that from fuel combustion.

An immediate and rough assessment of damages from urban air pollution in a particular location (or of the benefit from reduction of pollution), can be obtained by multiplying the numbers in Table 6.1 by the following values: (a) annual average exposure

Table 6.1 Suggested values of damages from (or benefits from reduction of) urban air pollution (thousands of U.S. dollars per $\mu g/m^3$ change in concentrations of local pollutants per 1 million people at US$1,000 per capita income)

1 $\mu g/m^3$ change in:	*Health cost*	*Cost of mortality*	*Cost of morbidity*	*Nonhealth cost*	*Visibility*	*Soiling*	*Corrosion*	*Total*
PM_{10}[a]	1,116	500	660	81	25	56		1,240
Sulfur dioxide (SO_2)	17		17	61			61	77
Nitrogen oxides (NO_x)				27			27	27

Note: Assumes a crude mortality rate of 8 per 1,000 population; share in population of children under 14, 27 percent; share in population of asthmatics, 5 percent (see Table 6.2). Also assumes the following income elasticities: for health effects and visibility, 1; for soiling, 0.9; and for corrosion, 0.65.

a. The pollution mix (the chemical and size composition of PM_{10}) is assumed to be typical of that of fuel combustion emissions.

Source: Authors' calculations.

to the respective pollutants (measured in μg/m^3); (b) the exposed population (measured in millions of people); and (c) average income (in thousands of U.S. dollars), with application of the relevant income elasticities.

Note that special care and adjustments should be used when applying the values for health damages from Table 6.1 to the range of PM_{10} concentrations based on ambient measurements if the range is high (over 150 μg/m^3) or if a substantial share of particles from nonfuel sources is present (see Chapter 3).

More accurate estimates will require modification of the mortality rate and the share of children in the population according to the local situation. Table 6.2 contains estimates of the health costs attributable to PM_{10} exposure for various typical values of these parameters, with all other assumptions as in Table 6.1. The selected values of the parameters provide a midrange estimate of the health costs (keeping all other variables constant). Two other parameters used by some dose-response functions—share of asthmatics in the population (for health costs) and current TSP levels (for visibility costs)—make no noticeable difference to the total costs of local pollution.

Damages per ton of local pollutants are difficult to generalize because they vary greatly across pollution sources and locations. Even when the height of the emission stack, as well as the characteristics of the exposed population, is taken into account, the impact of 1 ton of a given pollutant depends on the specific meteorological conditions in a given area. The difference in this impact is especially significant for small (low-stack) sources; the impact from modern power plants (high-stack sources) is more uniform. Table 6.3 illustrates the ranges and average values for the six cities of the health damages attributable to 1 ton of local air pollutants emitted from different sources (or the benefits from a reduction of 1 ton of pollutants). As in Table 6.1, all values are normalized on a population of 1 million with an average income of US$1,000 per person, assuming an elasticity of 1, a crude mortality rate of 8 per 1,000 population, and shares of children under 14 and of asthmatics of 27 and 5 percent, respectively.

Table 6.3 can be used as a quick reference guide to the plausible range of health damage estimates per ton of a local pollutant emitted from different combustion sources. (As with Table 6.1, the values in Table 6.3 should be adjusted for the exposed population and the income level in a particular study.) It should be kept in mind, however, that a significant error can be attached to the average values of damages derived from this table unless the values are validated at each step in the damage assessment method. The error can be

Table 6.2 Health costs of changes in particulate concentrations under various assumptions concerning mortality rate and age composition
(thousands of U.S. dollars per μg/m^3 change in PM_{10} per 1 million population at US$1,000 per capita income)

	Share of children under age 14				
Crude mortality rate per 1,000 population	*0.25*	*0.27*	*0.30*	*0.35*	*0.40*
6	1,045	1,034	1,019	994	969
7	1,107	1,097	1,082	1,056	1,031
8	1,170	1,160	1144	1,119	1,094
9	1,232	1,222	1,207	1,181	1,156
10	1,294	1,284	1,269	1,244	1,218

Note: Assumes share in population of asthmatics of 5 percent; income elasticity of 1.
Source: Authors' calculations.

Table 6.3 Health damages from (or benefits from reduction of) emissions of local air pollutants from various combustion sources, estimated in the six-city study
(U.S. dollars per ton of local pollutants per 1 million population at US$1,000 per capita income)

1 ton change in emissions of:	*High stack (modern power plants)*	*Medium stack (large industry)*	*Low stack (small boilers and vehicles)*
PM_{10}			
Range for six cities	20–54	63–348	736–6,435
Average	42	214	3,114
Sulfur dioxide			
Range for six cities	3–8	10–56	121–1,037
Average	6	33	487
Nitrogen oxides			
Range for six cities	1–3	3–13	29–236
Average	2	9	123

Source: Authors' calculations.

especially large for low- and medium-stack sources, as the wide ranges in damages for these sources demonstrate. It is therefore not advisable to use values from Table 6.3 when making a city-specific source apportionment of damages from low- and medium- stack sources, such as vehicles, residential stoves, and industrial and commercial boilers.

Robustness of the Health Cost Estimates

Table 6.4 shows the damage caused by combustion emissions of each local pollutant, both through a direct increase in the ambient levels of primary PM_{10}, SO_2, and NO_x and through formation of secondary sulfates and nitrates. The table illustrates that health costs amount to over 85 percent of total local pollution costs and that the health burden from PM_{10} emissions far exceeds all other items. Nearly 99 percent of the health costs is attributable to exposure to primary and secondary PM_{10}. Health damages from emissions of SO_2 and NO_x are more than twice

Table 6.4 Local damage from emissions of various pollutants: Six cities
(millions of 1993 U.S. dollars)

	Mumbai	*Shanghai*	*Manila*	*Bangkok*	*Krakow*	*Santiago*	*All six cities*	*Percent, by type*	*Percentage of total*
Health costs	*151*	*730*	*390*	*491*	*88*	*780*	*2,629*	*100*	*86*
From PM_{10} emissions	117	604	254	321	69	558	1,923	73	63
From sulfur dioxide emissions	20	107	65	64	15	102	374	14	12
From nitrogen oxide emissions	14	19	71	106	3	120	332	13	11
Nonhealth costs	22	96	*101*	74	9	*112*	*414*	*100*	*14*
From PM_{10} emissions	6	36	16	20	4	32	114	28	4
From sulfur dioxide emissions	9	47	37	19	4	26	142	34	5
From nitrogen oxide emissions	7	14	47	35	1	53	159	38	5
Total local damage	*173*	*826*	*491*	*565*	*97*	*891*	*3,044*	*100*	*100*

Note: Numbers may not sum to totals because of rounding.
Source: Authors' calculations.

the nonhealth damages caused by these emissions and are almost entirely the result of their contribution to secondary particulates.[17]

Health costs from exposure to PM_{10} clearly dominate the local environmental costs of fuel use. Our confidence in these estimates is therefore the key to validating the results of the analysis and the policy conclusions presented in Chapter 1 of this paper. The choice of assumptions underlying our calculations of health damages was extensively discussed in Chapters 3 and 4, where we also pointed to major uncertainties and alternative assumptions. Throughout the paper we have stressed that we are using a rather conservative approach to quantifying the health costs of air pollution. Table 6.5 summarizes these arguments by showing how our estimate of the health costs from a 1 $\mu g/m^3$ change in PM_{10} exposure relates to other estimates that could be derived by using alternative assumptions for the principal parameters.

In Table 6.5 the values of the health costs are compared with the *base* value (test 1), which is derived on the basis of the assumptions concerning dose-response and valuation parameters summarized in Tables 3.4 and 4.3, using an elasticity of 1 for income adjustment. Each test specified in the second column of Table 6.5 is performed under the condition that all the other assumptions, except for those being tested, are the same as for the base value. All estimates of health costs are normalized to the same situation as described above: a change in 1 $\mu g/m^3$ exposure to PM_{10} for a population of 1 million with an average per capita income of US$1,000, a crude mortality rate of 8 per 1,000 population, and a share of children in the population of 27 percent.

Most of the alternative assumptions are commonly used by other air pollution valuation studies or policy analyses, including analyses that underlie air quality regulations in the United States and the countries of the European Union. The tests are performed for the following main parameters or their combinations (see Chapters 3 and 4 for a more detailed explanation):

- Change in mortality risk (tests 2 and 3)
- Change in the risk of chronic bronchitis (tests 4 and 5)
- Value of a statistical life (tests 6–12)
- Income elasticity of willingness to pay (WTP) for reducing health risks (tests 13–16).

In addition, for estimates 17–19 no value is attached to mortality risk from exposure to ambient PM_{10}. Although this is not a plausible assumption, since evidence of the impact of exposure to particulates on mortality is very strong, it is given here for those who find the value of a statistical life (VOSL) concept too controversial. Note that tests 5, 8, 11, 12, 15, 16, 18, and 19 assign alternative values for two or more of the parameters. Estimate 20 is a simple average of all the other estimates and is intended to show the order of magnitude of the health costs emerging from collective evidence across major studies and assumptions.

All but two of the assumptions tested in Table 6.5 give a higher value of health costs than the base value used in this study. One of the assumptions that results in a lower value for the health costs (21 percent lower than the base value) is the use of half of the VOSL for air pollution–related death that was adopted in the study (test 10). This assumption reflects the most recent work on valuing latent mortality risks from exposure to air pollution (see, for example, Krupnick et al. 1999), as opposed to valuing immediate risks, as has been done in labor-market studies. Although this approach implies a significantly lower VOSL, it is to be applied with a much higher dose-response function for mortality, based on chronic exposure studies (see Chapter 3). When mortality risk estimates that take account of

Table 6.5 Variations in health costs under various assumptions (thousands of U.S. dollars per 1 $\mu g/m^3$ change in PM_{10} per 1,000,000 population at US$1,000 per capita income level)

Test number	*Assumptions*	*Health costs*	*Percentage change from base value*
1	*Base*	*1,159*	*0*
2	Mortality change of 1.94 percent	1,814	57
3	Mortality change of 4.2 percent	3,158	172
4	Central estimate for chronic bronchitis (6.12)[a]	1,433	24
5	2 and 4 combined	2,087	80
6	Base VOSL of US$3.6 without adjustment for DALYs	2,231	92
7	VOSL of US$4.8 million without adjustment for DALYs[b]	2,140	85
8	VOSL of US$4.8 million without adjustment for DALYs, plus 4[b]	2,596	124
9	VOSL of US$4.8 million with adjustment for DALYs	1,417	22
10	50 percent of the base VOSL after adjustment for DALYs[c]	910	–21
11	50 percent of the base VOSL after adjustment for DALYs, plus 2[c]	1,237	7
12	25 percent of the base VOSL after adjustment for DALYs, plus 3[c]	1,284	11
13	Income elasticity of 0.7	2,922	152
14	Income elasticity of 0.55	4,639	300
15	4 and 13 combined	3,612	212
16	4, 11, and 13 combined	3,807	228
17	No mortality risk valuation	660	–43
18	13 and 17 combined	1,663	43
19	4, 13, and 17 combined	2,353	103
20	Average of all estimates	2,164	87

Note: VOSL, value of a statistical life. The base value assumes a crude mortality rate of 8 per 1,000 population; share in population of children under 14, 27 percent; share in population of asthmatics, 5 percent; and income elasticity of 1.

a. The central estimate for chronic bronchitis, from Ostro (1994), is commonly used in a large number of studies.

b. The assumptions are from USEPA (1997). Note that the value for chronic bronchitis was adjusted for a higher VOSL (see Chapter 4).

c. Assumptions 10–12 reflect the most recent work on valuing latent mortality risks from exposure to air pollution (for example, Krupnick et al. 1999), as opposed to valuing immediate risks in labor-market studies. The former approach implies a significantly lower VOSL, but it is supposed to use a much higher dose-response function for mortality, based on chronic exposure studies.

chronic exposure are used, the health costs exceed the base value despite a significantly lower VOSL (see tests 11 and 12).

The other assumption that leads to lower health costs (43 percent lower than the base value) is obviously the one that includes no value for mortality risk (test 17). Interestingly, even with this unrealistically conservative assumption, the health costs exceed the base value when a lower income elasticity, supported by recent studies in developing countries, is used (see tests 18 and 19 and Chapter 4).

One of the most important observations from Table 6.5 is that even if the mortality costs, which are often surrounded by significant controversy and uncertainty, were ignored, it would not change the major qualitative conclusions presented in Chapter 1. The lowest health cost, as defined in test 17 (57 percent of the base value) would still dominate the environmental costs of fuels: the share of health costs in total environmental costs would be 55 percent of the total damage, compared with 30 percent for climate change costs and 15 percent for local nonhealth costs for the sample of six cities. The magnitude of damages, both in absolute terms and per ton of fuel, would still be extremely high, especially for small fuel users. Despite the downward effect of the excessively conservative assumptions, which are not consistent with epidemiological evidence or major valuation studies, the average estimate of health costs in Table 6.5 (test 20) is almost twice the base value.

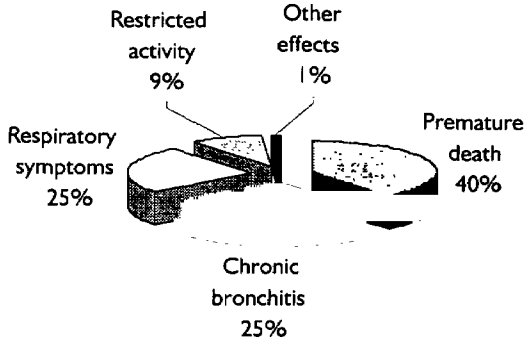
Major Areas for Further Research and Development

The health cost estimates appear to be sufficiently robust for the purposes of this study, which is a cross-country comparison aimed at establishing broad priorities for addressing environmental damages from fuel use. More detailed analyses of the costs and benefits of abatement strategies for a particular country or city will often require that the techniques be refined and the uncertainties of this rapid assessment be reduced.

In this context, it is important to test the impact on the ultimate outcome of the health assessment of key individual parameters or assumptions adopted at various steps of the assessment. This can help guide further research toward improving the accuracy of health cost calculations by focusing on the factors that matter most. Debate about arriving at better values for a particular parameter for health impact valuation often lacks this wider perspective. The result can be that the most attention is directed toward improving accuracy in the estimation of a particular parameter even though there is greater uncertainty about a more critical parameter.

For example, various epidemiological studies have obtained different estimates for mortality risk from exposure to PM_{10}, and there is no unanimous agreement on the choice of the best value. When end-points of air pollution exposure are reduced to a single denominator through the valuation exercise, however, premature deaths account for about 40 percent of the health costs, and various illnesses account for 60 percent (see Figure 6.1 and Tables 1.2 and 4.4). A larger share of morbidity costs is consistent with the estimates from other studies (for example, World Bank 1994, 1997a; USEPA 1997). Chronic bronchitis and acute respiratory symptoms are the largest contributors to the economic costs associated with morbidity. Note that the morbidity contribution of 60 percent is based on a low estimate for chronic bronchitis of 3.06 (see Chapter 3). If a central estimate of 6.12 were used, the morbidity outcomes would amount to 69 percent of the total health costs.

Figure 6.1 Composition of the health costs attributable to air pollution in the six cities, by cause



Source: Authors' calculations based on the assumptions of this study, 1993.

A sharper focus on the share of the social costs of sickness in the total health damages due to air pollution can be used to strengthen dialogue with policymakers, as it reduces reliance on arguments that are surrounded by the controversy over valuing a statistical life. But morbidity cost estimates, even though seemingly not as controversial as estimates for premature death, can be less accurate than mortality costs, for several reasons. First, despite the larger contribution of morbidity costs to total health costs, fewer air pollution studies have measured illness than have measured death, even when all illnesses are counted together. Second, many of the morbidity studies are relatively old, use less sophisticated techniques, and measure exposure to TSP rather than PM_{10}. In particular, commonly used dose-response coefficients for chronic bronchitis are based on a single study, published in 1993, that used TSP measurements. Third, only a limited number of health end-points related to air pollution has been quantitatively assessed. Fourth, better valuation techniques for acute and, especially, chronic illnesses linked to air pollution need to be developed. Again, there is by far less information and greater uncertainty about the value of chronic bronchitis than about the VOSL. Together, these facts point to the need for more studies to assess and value major morbidity outcomes.

As far as the effect of air pollution on mortality is concerned, there is a strong case for shifting the focus to measurement of the impacts of chronic exposure. Consistent evidence from the latest PM_{10}-based studies indicates that additional acute exposure studies are unlikely to dramatically change what is already known and will not provide full information about the magnitude of the impact. Prospective cohort studies, which provide the most accurate information on the impact of chronic exposure, are very expensive, take a long time, and require very careful design, implementation, and analysis. Although a study of this kind in a developing country could be an invaluable source of information that deserves support from the international community, it is unrealistic to expect several studies covering different countries to be done. Thus, study selection, methodology, and design should be subject to close scrutiny to ensure that the results of any such study yield information that can be used in various countries with sufficient confidence.

The importance of chronic exposure and latent effects implies the need to revisit the approach to valuing air pollution–related deaths by mastering valuation studies that capture people's preferences in trading off *future* risks. This in turn requires the development of techniques for translating the annual reduction in exposure (as a result of pollution control policies and measures) into annualized benefits from the respective changes in the risk of illness or death over a lifetime.

By far the greatest uncertainty affecting the valuation results is linked to the assumed relationship between income level and willingness to pay for reducing the risk of ill health. For example, a 10 percent increase in the base coefficients for mortality risk and chronic bronchitis raises the base health cost in table 6.4 by 4 and 2 percent, respectively, while a 10 percent increase in the WTP elasticity of 1 reduces the health cost by as much as 26

percent, and a 10 percent decrease in the assumed elasticity increases the health cost by 17 percent. Thus, a critical priority area for future work is to determine willingness to pay for avoiding morbidity and mortality impacts in developing countries and to generate stronger evidence on the income elasticity of willingness to pay across countries with very different income levels. In certain cases, aggregate measures such as DALYs that do not involve direct costing of the health effects attributable to air pollution can be used to rank impacts and mitigation options within a particular country (with a cutoff point in the cost per DALY saved typical for public health interventions in that country).

The discussion above has focused on improvements in quantifying and valuing health impacts from given levels of exposure. The rapid damage assessment method implemented in this study estimates levels of exposure on the basis of fuel and emissions inventories. Another important measure for improving the accuracy of the method for assessing damages from various pollutants, sources, and fuels is to refine atmospheric modeling and emissions factors. As discussed in Chapter 2, major development contributions will come from photochemical modeling of sulfates, nitrates, and ozone and from the determination of emissions factors for specific technologies used in developing countries such as the two-stroke motorcycles and three-wheel vehicles common in Southeast Asia.

Detailed analyses for specific cities will also benefit from a careful assessment of human exposure. Especially if the relative contributions of various pollution sources and fuel uses to health damage are at issue (and need to be understood in order to set priorities for urban air quality management), more effort in estimating actual levels of exposure is warranted. This would require taking into account such factors as how many people spend how much time in various areas of the city (for example, in municipalities with a different air quality situation, along major roads, and in residential areas, both indoors and outdoors) and matching these data with the contributions of various sources and sectors to pollution levels in those areas.

Finally, in further applications of this method, it should be kept in mind that there are significant uncertainties in the estimates of global and nonhealth local damages in this study and that the impacts of long-range (regional) pollution have been ignored. Existing studies suggest that these local and long-range nonhealth damages are typically much smaller than those to human health (see Chapter 5), although regional and country variations in the effects of long-range pollution can be substantial (see, for example, Downing, Ramankutty, and Shah 1997). Depending on the pollution situation in a particular location and the purpose of the analysis, the scope of some assessments can be limited to estimation of health damages. Assessments of environmental damages from fossil fuels in areas where long-range deposition is a serious problem (as in China) should attempt to include this type of damage.

The most difficult task is to arrive at plausible present-value estimates of damage from carbon emissions (and from other greenhouse gases), given the long time horizon and the great uncertainty about the range and magnitude of specific effects. Climate change is an area of active and evolving research. Although estimates of market impacts in industrial countries have tended to fall as investigators have improved their analysis, there is growing concern about nonmarket damages (e.g., on ecosystems) and the impact on developing countries (see, for example, Mendelsohn and Dinar 1999). This creates a new and important dimension in the climate change policy agenda. Future assessments of damages from climate change should keep abreast of ongoing research and use updated knowledge to adjust the damage values or modify the approach.

Annex A — Base Emissions Factors for Local Pollutants

In the proposed methodology, estimates of emissions are based on a citywide inventory of fuel use in different sectors and on standard emissions factors (see Table A.1).

and gasoline engines were adjusted for the specific fleet composition of the pools of diesel or gasoline vehicles, as these factors depend on the type and age of vehicles. For example,

Table A.1 Base emissions factors for major combustion processes

	Emissions factor (kg per ton fuel)			
Fuel and combustion process	*TSP*	*PM_{10}*	*SO_2*	*NO_x*
Coal				
Utility boiler	0.15 * 5 * A	0.6 * TSP em.f.	19.5 * S	10
Large industrial boiler	0.5 * 6 * A	0.5 * TSP em.f.	19.5 * S	7
Small industrial boiler	1.5 * A	0.5 * TSP em.f.	17.5 * S	4
Household boiler	1.5 * A	0.5 * TSP em.f.	15.5 * S	1.5
Fuel oil				
Utility boiler	0.15 * 2 * (0.38 + 1.25 * S)	0.9 * TSP em.f.	20 * S	8.5
Large industrial boiler	2 * (0.38 + 1.25 * S)	0.8 * TSP em.f.	20 * S	7
Small industrial boiler	3 * (0.38 + 1.25 * S)	0.8 * TSP em.f.	20 * S	4.5
Household boiler	3 * (0.38 + 1.25 * S)	0.8 * TSP em.f.	20 * S	2.7
Wood boiler	14	0.5 * TSP em.f.	0.02	1.7
Diesel engine	15	0.8 * TSP em.f.	20 * S	40.5
Gasoline engine				
No catalytic converter	0.82	0.9 * TSP em.f.	0.45	42.5
Three-way catalytic converter	0.8	0.9 * TSP em.f.	0.45	9.6

Note: A, ash content of coal, weight percent; S, sulfur content of fuel, weight percent; TSP, total suspended particulates; em.f., emissions factor. Assumptions on city-average level of pollution control: 85 percent removal efficiency for TSP control at coal- and oil-fired power plants; 50 percent removal efficiency for TSP control at large industry; no controls for other pollutants and sectors. Emissions factors for fuel oil boilers and diesel engines assume that equipment is not new and not well maintained.

Sources: USEPA (1986); WHO (1989); authors' assumptions on pollution control and PM_{10}/TSP emissions ratios.

Default emissions factors and assumptions were refined for the six cities where locally or regionally specific information was available. Data permitting, emissions factors for diesel

emissions factors for gasoline engines for Bangkok, Manila, and Mumbai attempt to take account of a large number of two-stroke engines (two- and three-wheelers).

Annex B — The Dispersion Model

The emissions-concentrations relationship for low-, medium-, and high-stack sources are as follows. The change in ambient concentrations of carbon (in $\mu g/m^3$) per unit emissions from source h (where emissions are measured in metric tons and h can be either high, medium or low) is:

$$\Delta C = D^h / 10{,}000 \qquad \text{(B.1)}$$

where the dispersion coefficients D^h for low-stack sources are given by:

$$D^h = \sum_{km} f_{km} \exp[\alpha^h_{km} + \beta^h_{km} \times \ln(R)] \qquad \text{(B.2)}$$

and for medium- and high-stack sources are given by:

$$D^h = \sum_{km} f_{km} \exp(\alpha^h_{km} + \beta^h_{km} x \ln(R) + \gamma^h_{km} \times [\ln(R)]^2\}. \qquad \text{(B.3)}$$

R is the radius of the city, computed by the formula:

$$R = \sqrt{A/\pi} \qquad \text{(B.4)}$$

where A is the area of the city in square kilometers (the city need not be circular). The parameters α^h_{km}, β^h_{km}, and γ^h_{km} are taken from Tables B.1, B.2, and B.3. The frequency parameters f refer to the contents of Table B.4.

If data on wind speed and atmospheric stability are not available for the area under consideration, an approximate estimate of ground-level concentrations can be made using the dispersion coefficients for areas with similar meteorological characteristics, suitably adjusted for the size of the city. If no meteorological data exist and no dispersion parameters are available from similar areas, a very rough estimate of the dispersion parameters can be obtained purely as a function of the diameter of the city by using a set of default parameters. Note that derivation of the appropriate meteorological data may involve processing several years of hourly meteorological readings.

Table B.1 Coefficients for the dispersion model: Low-stack (or low-level) area sources

Atmospheric stability and wind speed (miles per second)	α	β
Unstable		
< 2	6.391	−1.492
2–5	6.075	−1.724
5–7.5	5.925	−1.712
> 7.5	5.800	−1.682
Neutral		
< 2	7.778	−1.592
2–5	6.843	−1.600
5–7.5	6.245	−1.619
> 7.5	5.893	−1.624
Stable		
< 2	7.398	−0.872
2–5	7.256	−1.241
5–7.5	6.976	−1.433

Sources: Dennis (1978); WHO (1989); Sebastian, Lvovsky, and de Koning (1999).

Table B.2 Coefficients for the dispersion model: Medium-stack point sources

Atmospheric stability and wind speed (miles per second)	α	β
Unstable		
< 2	6.391	−1.492
2–5	6.075	−1.724
5–7.5	5.925	−1.712
> 7.5	5.800	−1.682
Neutral		
< 2	7.778	−1.592
2–5	6.843	−1.600
5–7.5	6.245	−1.619
> 7.5	5.893	−1.624
Stable		
< 2	7.398	−0.872
2–5	7.256	−1.241
5–7.5	6.976	−1.433

Sources: Dennis (1978); WHO (1989); Sebastian, Lvovsky, and de Koning (1999).

Table B.3 Coefficients for the dispersion model: High-stack point sources

Atmospheric stability and wind speed (miles per second)	α	β	γ
Unstable			
< 2	1.171	0.855	−0.284
2–5	1.034	0.427	−0.230
5–7.5	0.600	0.327	−0.216
> 7.5	0.647	0.251	−0.232
Neutral			
< 2	−30.801	19.537	−3.117
2–5	−13.820	7.981	−1.226
5–7.5	−9.381	6.243	−1.124
> 7.5	−6.275	4.250	−0.821
Stable			
< 2	−18.380	3.978	0.000
2–5	−44.510	20.894	−2.654

Sources: Dennis (1978); WHO (1989); Sebastian, Lvovsky, and de

Table B.4 Example of a completed meteorological frequency factor table

Atmospheric stability and wind speed (miles per second)	*Frequency factor (f)*
Unstable	
< 2	$f_{11} = 0.0$
2–5	$f_{12} = 0.0$
5–7.5	$f_{13} = 0.1$
> 7.5	$f_{14} = 0.1$
Neutral	
< 2	$f_{21} = 0.1$
2–5	$f_{22} = 0.2$
5–7.5	$f_{23} = 0.2$
> 7.5	$f_{24} = 0.1$
Stable	
< 2	$f_{31} = 0.1$
2–5	$f_{32} = 0.1$
5–7.5	$f_{33} = 0.0$

Note: Frequency factors must sum to unity.
Sources: Dennis (1978); WHO (1989); Sebastian, Lvovsky, and de Koning (1999).

Calibration of the Model

In this context, calibration means adjusting the results of the model such that the predicted concentrations of the various pollutants in the modeling results match the concentrations actually measured. Calibration is highly desirable; even in the most sophisticated of dispersion models, there can be a large discrepancy between predicted and actual concentrations. Ideally, calibration of the dispersion model described above requires both an emissions inventory of the city itself and a measure of background (or rural) concentrations. The procedure is first to define and then to adjust a scaling parameter in the equation linking emissions and concentrations such that, given the emissions inventories for particular pollutants, the model reproduces exactly the increase in ambient concentrations above background levels. If a complete emissions inventory and sufficient data on background levels of pollution are not available, calibration should at least ensure that the

computed concentrations arising from fuel use do not exceed concentrations of those pollutants based on ambient measurements and are in a range of a plausible proximity to measured values. In this analysis the model was calibrated for SO_2, ambient levels of which were assumed to be largely caused by fuel combustion within a city. For Krakow and Shanghai the assumption was that about 20 percent of SO_2 comes from outside sources (large power plants).

Modeling Secondary Sulfates and Nitrates

The process of formation of secondary sulfates and nitrates is complex, and an accurate approach requires sophisticated dispersion/photochemical modeling that is beyond the capacity of a rapid assessment exercise. The approach taken here was to review and draw on available evidence (based on chemical analyses of the ambient air or on appropriate dispersion modeling) on the species composition of fine particulates in various locations. Secondary sulfates and nitrates (not emitted as such but formed in the air from SO_2 and NO_x emissions) are functions of the ambient levels of these pollutants. A modeling of the Shanghai area estimated that average concentrations of sulfates are about one-sixth of average SO_2 concentrations (Streets et al. 1997).

While we recognize that this relationship will vary across locations, depending on climate and meteorological conditions, we took a very primitive approach in our calculations. Sulfate levels were assumed to be 16 percent of the estimated SO_2 levels, and, after testing a number of assumptions, levels of secondary nitrates were modeled at 4 to 6 percent of the estimated NO_x concentrations in the individual cities in question. For each city, these crude assumptions were validated by the analysis of data on the levels of sulfates and nitrates in various places in relation to the composition of combustion sources, to ensure that the resulting concentrations of sulfates and nitrates would fall in a plausible range for a given city based on testing and cross-checking several parameters. In the version reported in this paper, shares of secondary sulfates in fuel-induced PM_{10} vary across cities from 13 to 17 percent; shares of secondary nitrates have a larger range, from 3 to 22 percent. The contribution of sulfates to PM_{10} is greatest in Krakow, where large volumes of coal are used, whereas nitrates make the highest contribution to PM_{10} in Bangkok. Overall, the contribution of secondary particulates to exposure to PM_{10} from fuel use for the whole sample of six cities is 25 percent. That figure is the share of secondary sulfates and nitrates in incremental PM_{10} levels that is attributed to fuel combustion only; it does not represent the ambient level observed in these cities. This estimate seems to provide a reasonable, and rather conservative, proxy that is believed to be a better alternative than simply ignoring the contribution of secondary particulates.

It should be emphasized that the purpose of this exercise is to estimate the damage costs from fuel combustion, not the levels of secondary particulates per se. Thus, the robustness of the approach should be evaluated in the context of the overall assumptions and results of this rapid assessment. Note that the PM_{10}-based dose-response functions were applied to the estimated contributions of secondary sulfates and nitrates, not the existing dose-response functions for sulfates and $PM_{2.5}$, which are much higher. Thus, the effective contribution of secondary particulates to the health effects and damage costs of fuel use was assumed to be significantly lower than the percentages of PM_{10} given above. This means that the modeled contribution of secondary particulates to health damage from PM_{10} exposure is at the lower end of the possible contribution and does not overstate the social costs of fuels.

Annex C — Estimating Predicted Willingness to Pay (WTP) to Avoid Morbidity

In TER (1996), WTP estimates are determined for each of several health states. A translog model is fitted to the data, and it can be shown that the following simplification is acceptable:

$$\ln(WTP) = 3.68 + 2.75 * [\ln(QWB)]^2 - 1.55 * \ln(QWB) * \ln(DAYS) \quad \text{(C.1)}$$

where *WTP* is willingness to pay in 1993 U.S. dollars; *QWB* is quality of well-being; and *DAYS* is the duration of the illness in days. The estimated regression has an R^2 of 0.89 and, in keeping with the results of Brajer et al. (1991), there appears to be an interaction between the QWB score and the duration of the illness.

This equation is next used to predict willingness to pay to avoid health states relevant to the air pollution epidemiology literature that are not dealt with in the morbidity valuation literature. To do so, the QWB score corresponding to each health state and a statement regarding the duration of the illness are required (Table C.1). The predicted WTP for the morbidity end-points related to air pollution are given in Table

Table C.1 Derivation of quality of well-being (QWB) scores for different health states identified in the air pollution literature

Health status	*Mobility*	*Physical activity*	*Social activity*	*Symptom*	*QWB score*[a]
Respiratory hospital admissions	0.090	0.077	0.106	0.299	0.428
Asthma attacks	0	0.060	0.061	0.257	0.622
Emergency room visits	0.062	0.077	0.061	0.299	0.501
Bed disability days	0	0.077	0.061	0.257	0.605
Lower respiratory illness in children	0	0	0	0.257	0.743
Respiratory symptoms	0	0	0	0.257	0.743
Cough days	0	0	0	0.257	0.743
Chest discomfort days	0	0	0	0.299	0.701
Minor restricted activity days	0	0	0	0.257	0.743
Eye irritation	0	0	0	0.230	0.770
Phlegm	0	0	0	0.170	0.830

Note: a. The QWB score is 1 minus the sum of the weights in the preceding columns.
Source: Kaplan et al. (1993).

C.2. Restricted activity days (RADs) are defined as a weighted average of a bed disability day (BDD) and a minor restricted activity day (MRAD), employing the weights 0.4 and 0.6, respectively.

Table C.2 Willingness to pay to avoid health states identified in the air pollution literature

Health status	*QWB score*	*Duration (days)*	*Predicted WTP (1990 U.S. dollars)*
Respiratory hospital admissions (RHAs)[a]	0.428	9.5	4,275
Asthma attacks	0.622	1	63
Emergency room visits	0.501	1	126
Bed disability days (BDDs)	0.605	1	69
Lower respiratory illness in children, cases	0.743	1	44
Respiratory symptoms	0.743	1	44
Cough days	0.743	1	44
Chest discomfort days	0.701	1	50
Minor restricted activity days (MRADs)	0.743	1	44
Eye irritation[b]	0.770	1	41
Phlegm[b]	0.830	1	38
Restricted activity days (RADs)	—[c]	1	53

Notes: a. Average duration of RHAs refers to average duration of admissions due to emphysema and bronchitis (National Heart, Lung and Blood Institute, National Institutes of Health, Bethesda, Md.).
b. Values for eye irritation and phlegm were not used in this six-city exercise but are given for further applications of the approach.
c. RADs are a weighted average of BDDs and MRADs.
Sources: TER (1996); authors' calculations.

Annex D — Values for Visibility, Soiling, and Corrosion

Visibility

It is important to distinguish between the benefits of visibility in the cities where people live and work and in, for example, national parks, where the purpose of a visit is mainly recreational sightseeing. This section deals exclusively with the former set of benefits. It is perhaps not valid to suppose that since individuals spend most of their time at home or at work, residential values are of greater importance than recreational values. Nevertheless, the dispersion model used in this report deals exclusively with air pollution within the urban area itself (see Annex B), as the extent to which emissions arising from particular urban conurbations degrade visibility in particular areas of major recreational interest cannot be known. Furthermore, even if because of the aesthetic qualities of nearby national parks, visibility possesses greater value in that context, it is difficult to see how the existing literature relating to visibility in the Grand Canyon National Park (Balson, Carson, and Mitchell 1990) or the national parks of the southwestern United States in general (Schulze et al. 1983) could easily be adapted for use in the context of other recreational areas, each of which is unique.

Information on visibility ranges in urban areas of developing countries and on how they might be affected by differences in pollution is not readily available. Nonetheless, there exists a close relationship between the concentration of particles, their light-scattering coefficient, and visibility. The nature of this relationship in the context of urban airsheds is discussed by Noll et al. (1968). The meteorological range of visibility is inversely proportional to the scattering coefficient of the particles (assuming that the absorption of light by particles and gases is insignificant). If it is assumed that the scattering coefficient is proportional to the mass of particles per cubic meter of air, a very tractable relationship exists. This last step is a simplification, since the scattering coefficient is a function not only of the total mass of particles but also of their size.

Notwithstanding the problem of differences in the sizes of particles, the following relationship, attributable to Noll et al. (1968), has been tested and has been found to provide a satisfactory approximation of prevailing visibility in a variety of locations:

$$V = \frac{884.8}{M} \qquad \text{(D.1)}$$

where V, measured in miles, is prevailing visibility (defined as the greatest visibility that is attained or surpassed around at least half of the horizon circle, not necessarily in continuous sectors, at noon in dry conditions) and M is the concentration of particulate matter in $\mu g/m^3$. Note that in this relationship, as pollution increases, the marginal effects on visibility decrease. This point is extremely important in what follows. Also, the relationship is dependent on ambient humidity being below 70 percent of the saturation point, to eliminate the possibility of any reduction in visibility due to

adsorption on particles. More complex relationships linking visibility range to particulates of various descriptions, absorption by gases, and humidity levels are also available (see, for example, Landrieu 1997).

The literature detailing willingness to pay for improvements in visibility in the context of the home and workplace has generally utilized the contingent valuation method (CVM) approach, whereby respondents are shown photographs relating to different visual ranges and are asked their willingness to pay for changes in the frequency with which particular visual ranges prevail. Unfortunately, under this approach it has been found difficult to distinguish between willingness to pay for improvements in visibility and other motivations such as health benefits, reduced soiling, and so on. The evidence is limited because of the small number of studies that have accounted for the possible embedding of other perceived benefits aside from enhanced visibility. This is a significant criticism that has undermined confidence in the results of previous major studies (e.g., Tolley et al. 1984). McClelland et al. (1991) sought to overcome the embedding problem by asking respondents to allocate their expressed WTP to different motivations. Only 18 percent of the expressed willingness to pay for the improvement in air quality depicted in that study was ascribed to the motivation of improving visibility.

The existing literature comprises five separate studies (Brookshire et al. 1979; Loehman et al. 1980; Rae 1983; Tolley et al. 1984; McClelland et al. 1991) and incorporates nine different U.S. cities. The results are analyzed by means of an unweighted least-squares regression. The following equation was estimated to explain household willingness to pay. Note that the functional form is selected such that the willingness to pay for a zero change in visibility is zero and that the equation is consistent with the assumption of diminishing marginal willingness to pay for visibility enhancement. The estimated equation is:

$$WTP = 46.4 \times \ln(V_2/V_1) + 79.7 \times DUMMY \times \ln(V_2/V_1) \qquad \text{(D.2)}$$

where *WTP* is individual willingness to pay in 1993 US dollars; V_1 and V_2 are the initial and final visibility ranges measured in miles; and *DUMMY* is a dummy variable that takes the value unity for the studies of Tolley et al. (1984) and zero otherwise. The equation explains 74 percent of the variation in the data. The significance of the multiplicative dummy variable illustrates the fact that the study of Tolley et al. suffers from significant embedding and should almost certainly be discounted (i.e., the last term in the equation should be dropped).

Unusually for damage cost estimates, the marginal damage curve falls as the concentration of particulates rises. The reason is that even though marginal willingness to pay for the additional mile rises as the current visibility range falls, the increase in the visibility range for a unit reduction in particulate concentrations falls even faster. This phenomenon has been noted by others (see Repetto 1981). The declining marginal damage function for impairment of visibility because of particulates may help explain why individuals living in large urban conurbations in developing countries might be unwilling to pay much at the margin for a reduction in air pollution insofar as its effect on visibility range is concerned: in a high-pollution situation, the improvement in visibility obtained from a unit reduction in particulate concentration is likely to be rather small (see Table D.1).

Although it is possible to present estimates of U.S. households' marginal willingness to pay for improvements in visibility, serious problems remain in transferring these estimates to the developing-country context. This is because the income elasticity of willingness to pay for improvements in the range of visibility is not known. However, it is probable that the income

Table D.1 Marginal willingness to pay for visibility improvements by restricting particulate concentrations, United States (1990 U.S. dollars)

Total suspended particulates (TSP) ($\mu g/m^3$)	*Visibility range (miles)*	*$\partial v/\partial m$ (miles)*	*$\partial WTP/\partial m$ (U.S. dollars per $\mu g/m^3$ TSP)*
50	17.7	0.35	0.80
100	8.8	0.09	0.50
150	5.9	0.04	0.30
200	4.4	0.02	0.20
250	3.5	0.01	0.10

Note: Damage cost estimates are given per $\mu g/m^3$ of TSP. To convert to PM_{10}, multiply by 1.81.
Source: Authors' calculations.

elasticity of WTP would be no less than unity, since the benefits from extended visibility are primarily aesthetic in nature. This assumption, coupled with the arguments given above, indicates that visibility effects are likely to be very small in developing countries.

Soiling

Air pollution results in soiling of materials and an increased need for washing and maintenance. Soiling in this context refers to particulate soiling within the household sector, not to soiling damage to the commercial and industrial sector or damage done to public buildings or historic monuments.

To place the alternative studies in proper context, it is instructive to consider briefly the theory underlying cleaning cost studies. In the most basic model the individual is modeled as consuming two goods: a numeraire good and a good referred to as cleanliness. Cleanliness is positively related to the frequency of cleaning and negatively related to the level of pollution. Each cleaning episode incurs a cost, and the individual has to allocate his or her budget between cleaning and the numeraire good. Within the context of this very simple model, it can be shown that the appropriate measure of marginal willingness to pay to reduce pollution is the increased expenditure required to maintain the current level of cleanliness. Furthermore the same model illustrates why the observed increase in expenditure on cleaning understates the true marginal willingness to pay. Given the choice, an individual confronted with an increase in pollution will note the increase in the relative price of cleanliness and purchase less of it. Thus, compensating the individual by the observed increase in expenditure on cleaning is insufficient. Note further that a reduction in cleaning expenditures is even acceptable as an economic outcome. A negative damage cost estimate as a lower bound is not very useful for a figure that is presumed to be positive.

This point is important because although several studies have been conducted on the effects of air pollution and soiling, most of them present increased expenditure on cleaning costs as the appropriate measure. Thus, for example, Ridker's (1967) study of how laundry and dry-cleaning costs vary across 144 U.S. cities with various levels of pollution could only provide a lower bound for the true WTP. Ridker justified his finding that no relationship between particulates and such expenditures existed by arguing that such operations are undertaken on a rigid schedule that is independent of location. In fact, however, such a finding is consistent with utility maximization.

The theoretically correct measure of WTP can only be derived from household production function (HPF) studies in which the observed expenditure on cleaning commodities is explained as the outcome of some utility maximization process. A significant criticism of the HPF models, however, is that although the expenditure surveys on which they are based include spending on cleaning outlays, they do not currently account for the time cost incurred by those who choose to do the cleaning themselves, since such information is not typically available in consumer expenditure

surveys. In fact, cleaning is likely to involve a considerable amount of own-labor input. According to Watson and Jaksch (1982), only a small part of all cleaning tasks is contracted out, and if labor costs, valued at typical contractual rates, were added, aggregate cleaning expenditures would rise by 400 percent. Thus, if only expenditures on cleaning products are accounted for, and not nonmarketed labor, the largest element of the costs associated with soiling from air pollution is omitted. Although it is questionable whether full market rates should be charged for cleaning tasks, this point still carries considerable force. Given this current shortcoming of HPF studies, Ridker's (1967) estimate might seem more appealing: define a number of different cleaning tasks, examine how the frequency with which they are undertaken varies according to ambient levels of particulate pollution, and then, on the basis of the cost of each task (including the costs of both labor and cleaning materials), compute the additional costs attributed to air pollution.

MRI (1980) is based on this approach. The frequency of 27 different cleaning tasks was determined by questionnaire for residents of Pennsylvania. Of these tasks, 11 proved sensitive to the level of air pollution, although data for only 9 of them were included in the actual report. The frequency of cleaning was linked in a regression to the ambient level of TSP. The associated costs were obtained from a survey of cleaning contractors.

The results of the MRI report are reproduced in Table D.2. They indicate a lower bound on soiling costs of US$2.82 per household per $\mu g/m^3$ of TSP in 1990 prices, or US$1.07 per capita. As anticipated, these exceed the measures obtained from the work of Watson and Jaksch (1982), who calculate a household WTP on the basis of the HPF approach. According to their study, an improvement in air quality from primary to secondary levels (a change of 15 $\mu g/m^3$ of TSP) results in a gain of US$25 per household in 1971 prices. Converting to 1990 prices gives US$5.40 per household per $\mu g/m^3$ of TSP, or US$2.10 per capita. These results can be compared with the CVM approach of McClelland et al. (1991), described in the preceding section. In principle, that study measures WTP directly and appears to indicate a household WTP of US$2.60 per $\mu g/m^3$ of TSP in 1990 prices, or US$1.00 per capita (although it is questionable whether this is a good way of eliciting preferences for avoiding soiling damages). The dissimilarity of the MRI results and the Watson and Jaksch results indicates that the former might not provide a very useful lower bound. The large difference between the McClelland et al. study and the Watson and Jaksch study, both of which purport to measure WTP, may reflect the strong assumptions that underlie the specification of the model used by Watson and Jaksch.

Once more, it is difficult to transfer these figures into the context of developing countries because of lack of information on the relevant income elasticities (that is, the willingness of poor individuals to take time from earning money to clean their houses) but also perhaps because of differences in housing, cultural values, and attitudes toward cleanliness. In this study we started with a median estimate of US$1.07 per individual per $\mu g/m^3$ of TSP and reduced it by half to adjust roughly for a typical urban household in a developing country, for which the list of tasks in Table D.2 is clearly excessive. The value can be converted to a PM_{10}-based measure by multiplying by 1.8.

Cross-sectional analysis of per capita expenditures on household cleaning goods in the 1980 International Comparisons Project (ICP) points to an income elasticity of demand for such commodities of 0.89, with a standard error of 0.09. This same income elasticity was used to scale estimates of soiling damage in this study.

Table D.2 Household cleaning tasks, their frequencies per unit of TSP, and their costs

Task	*Unit cost (1990 U.S. dollars)*	*Change in frequency per μg/m³ TSP*	*Additional cost (U.S. dollars per μg/m³ TSP)*
Replace air conditioner filters	3.38	0.005	0.02
Wash floors	20.29	0.040	0.81
Wash windows on inside	1.69	0.078	0.13
Clean venetian blinds/shades	11.84	0.048	0.57
Clean screens	0.67	0.006	0.00
Wash windows on outside	5.07	0.053	0.27
Clean storm windows	6.76	0.015	0.10
Clean outdoor furniture	33.82	0.006[a]	0.20
Clean gutters	50.73	0.014	0.71
Total			2.82

Note: The typical household has 2.63 persons. Prices were converted from 1970 to 1990 price levels using a consumer price index (CPI) deflator of 3.37. To convert from TSP to PM_{10} multiply by 1.8. Half of the total value is used in study calculations.
a. The frequency of cleaning outdoor furniture is presented as a nonlinear function of TSP concentrations. For the sake of simplicity, the figure given refers to the marginal effect evaluated at 100 μg/m³ TSP.
Sources: MRI (1980); authors' calculations.

Materials Damage

Air pollution–induced damage to materials, including stone, brick, painted surfaces, metals, rubber, and fabrics, is a widespread problem. The main pollutant involved is SO_2, but the corrosive effects may be reinforced by exposure to nitrogen dioxide (NO_2), ozone (O_3), and acidity in precipitation (H^+), as well as by the more general effects of the climate. Annual surface recession of exposed materials is predicted from the extensive international work on the relevant dose-response relationships (see Pearce 1997 for a recent survey). Most of the work on valuing material damage costs has utilized this dose-response literature.

The dose-response function cannot be used to compute economic damage directly; the next stage is to derive a damage function. As corrosion increases, the time that elapses before replacement, repainting, or repair takes place is shortened. This time lapse is determined by the rate of corrosion and the "acceptable" degree of damage. The concept of an acceptable degree of corrosion lends some indeterminacy to the establishment of damage functions, but in principle, this concept can be observed from actual behavior. Using such assumptions, it is possible to convert the dose-response functions to damage functions expressed as lifetime functions, that is, functions expressing the lifetime of materials as a function of exposure to pollutants. Although the total replacement cost may be constant by type of material, the annual cost of replacement will obviously be higher the shorter the time period to replacement; for any no-pollution context, there will be a given annual replacement cost, and there will be a higher annual replacement cost for the with-pollution case and so on for a gradient of levels of pollution. The difference between the two is the annual economic damage done by pollution.

Given these lifetime functions, valuation then requires the compilation of an inventory of materials exposed. This is a major task in itself without which the results of the dose-response-function literature cannot even be applied. After this, valuation of unit impacts is required. This valuation has generally taken the form of contractual costs for various maintenance jobs (i.e., the cost of materials plus the cost of labor). Reduced lifetimes of building materials or shortened maintenance cycles may not, however, fully reflect the true costs of air pollution; they ignore, among other things, avertive behavior such as use of corrosion-resistant materials, which can be a significant additional cost factor. In effect, the replacement cost approach to materials damage suggests that annual costs (*AC*) imposed by a change in the level of pollution are given by:

$$\Delta AC = UC \times SAR \times [\frac{1}{L_0} - \frac{1}{L_1}] \qquad \text{(D.3)}$$

where *UC* is the unit cost of replacement; *SAR* is the stock at risk; and L_0 and L_1 are the old and new levels of pollution, respectively. These lifetimes are themselves ratios of the critical (replacement) damage threshold to the annual amount of damage as indicated by the dose-response function.

This methodology assumes that the individual will take action at the same level of damage as before and will not attempt to maintain his or her property in a higher state of repair than previously. In fact, that assumption will lead to an overestimate of cost savings associated with maintenance but an underestimate of the overall economic benefits. More specifically, the connection between the damage-cost measure in the maintenance cycle approach and the conceptually more appropriate WTP measure is best understood by considering the market for maintenance services (see, for example, MATHTEC 1984).

Suppose that all householders and businesses have maintenance contracts with contractors to maintain their property at an agreed standard of repair. The price of maintaining property at a certain standard of repair is obviously an increasing function of the level of air pollution. If the level of air pollution falls, the cost of providing the same standard of maintenance also falls, and cost savings result. Indeed, these are the cost savings indicated by the lifetime maintenance approach. But this is not the end of the story because, unless the demand for maintenance services is completely price inelastic, the demand for maintenance services per time period will increase. Thus, the total change in economic surplus is given by the reduction in the cost of providing the old quantity of maintenance plus the surplus generated by the additional maintenance services demanded. Hence the cost savings on their own yield a lower bound on true WTP.

Calthrop (1996) has assembled the U.S. and European literature to derive estimates of damage per ton of pollutant (see Table D.3). Since methodologies differ and data reliability varies substantially, it is difficult to compare the results across studies. However, some general conclusions emerge: for the United States, a figure of about US$200 per ton of SO_2 seems consistent with the studies, with a range of US$150–$250 per ton. For Europe the range is wider, US$45–$2,020 per ton of SO_2, but with a range of US$250–$600 appearing more likely after outliers have been discounted. (All values are in 1993 dollars.) Finally, the values for NO_x are suspect, since the links between NO_x and damage to buildings are very uncertain and the studies are few.

Per capita averages of these damage costs are US$12.50 for SO_2 and US$11.30 for NO_2 or US$0.50 per $\mu g/m^3$ for SO_2 and US$0.25 per $\mu g/m^3$ for NO_2 in 1993 dollars. Conversion of these values to 1990 dollars yields US$0.45 for SO_2 and US$0.20 for NO_2. The latter is used in our calculations.

We turn, finally, to the task of transferring the results of this literature to the developing-country context. Pearce (1997) points out that use of damage cost figures from industrial countries takes no account of variations in the "acceptable" degree of damage or significant differences in the likely amounts of property per person. In particular, it is unreasonable to assume that the "critical threshold" is anything other than the outcome of choice and easy to envisage that what passes for a critical threshold in developing countries might be considerably in excess of what would be tolerated in the countries listed in Table D.3. Both assumptions ensure that spending on home repairs and maintenance rises as per capita income increases. The income elasticity of expenditure on repairs and household maintenance is calculated, using data from the 1980 International Comparisons Project, to be 0.64, with a standard error of 0.10. Scaling the damage cost estimates using the income elasticity of expenditure on household repairs and maintenance yields a lower-bound estimate for per capita costs of materials damage in other countries.

Table D.3 Estimates of costs of damage from corrosion: Various studies (1993 U.S. dollars)

Pollutant and country (various studies)	*Damage costs (dollars per ton)*	*Total emissions (thousands of tons)*	*Population (millions)*	*Damage per capita (dollars)*
Sulfur dioxide (SO_2)				
United States	150–250	17,700	250	14.20
United Kingdom	285	3,755	57	18.80
United Kingdom	45–250	3,755	57	9.70
United Kingdom	610	3,755	57	40.20
Germany	1,160–2,020	961	79	19.30
Germany	300–570	961	79	5.30
Germany	400	961	79	4.90
Germany	110	961	79	1.30
France	300	1,261	57	6.60
Average				12.50
Average per $\mu g/m^3$ SO_2				0.50 /pers/$\mu g/m^3$
Nitrogen dioxide (NO_2)				
United Kingdom	425	2,729	57	20.30
Germany	75	2,688	79	2.60
Average				11.30
Average per $\mu g/m^3$ NO_2				0.25 /pers/$\mu g/m^3$

Note: Annual average SO_2 concentrations are between 20 $\mu g/m^3$ and 30 $\mu g/m^3$ in the United States and the United Kingdom; average NO_2 concentrations are between 40 $\mu g/m^3$ and 50 $\mu g/m^3$ (for the urban areas that account for the most of the population and property values in those countries). The same concentrations are assumed for France and Germany. Thus, average damages per person were divided over 25 $\mu g/m^3$ for SO^2 and 45 $\mu g/m^3$ for NO_2.

Sources: Compiled by D. Maddison from Fisher, Chestnut, and Violette (1989); WEC (1992); Feliu, Morcillo, and Feliu (1993); Kucera et al. (1993, 1995); ECOTEC Ltd. (1994); Lipfert (1989, 1994); United Kingdom (1994); EC (1995); ApSimon and Cowell (1996); Calthrop (1996); Cowell and ApSimon (1996); Glomsrød et al. (1996); Haagenrud and Henriksen (1996); Kucera (1996); U.S. Bureau of the Census (1996); Landrieu (1997); WRI (1996); U.S. Department of Energy data.

Annex E — City Data on Fuel Use

Table E.1 Bangkok: Quantity and quality of fuel use, by economic sector

Fuel	*Modern power plants (H)*	*Suboptimal plants/ district heating (M)*	*Large industry (M)*	*Small industry (L)*	*Residential (L)*	*Land transport (L)*
Coal						
Thousands of tons						
Ash content (percent)						
Sulfur content (percent)						
Petroleum products						
Fuel oil						
Thousands of tons			2,300.0	500.0	300.0	
Sulfur content (percent)			1.5	0.5	0.25	
Motor diesel oil						
Thousands of tons						600.0
Sulfur content (percent)	0.5		0.5	0.5		0.5
Gasoline						
Thousands of tons						830.0
Fuelwood						
Thousands of tons						

Note: H, high-stack source; M, medium-stack source; L, low-stack (or low-level) source.
Sources: WHO and UNEP (1992): Radian International consultant report(1997).

Table E.2 Krakow: Quantity and quality of fuel use, by economic sector

Fuel	*Modern power plants (H)*	*Suboptimal plants/ district heating (M)*	*Large industry (M)*	*Small industry (L)*	*Residential (L)*	*Land transport (L)*
Coal						
Thousands of tons	3,096.0	175.0	12.0	143.0	84.0	
Ash content (percent)	11.8	12.0	12.0	9.9	12.0	
Sulfur content (percent)	1.5	1.5	1.5	1.5	1.5	
Petroleum products						
Fuel oil						
Thousands of tons				3.0		
Sulfur content (percent)				2.2		
Distillate oil						
Thousands of tons						
Sulfur content (percent)						
Motor diesel oil						
Thousands of tons						114.0
Sulfur content (percent)						0.3
Gasoline						
Thousands of tons						135.0

Note: H, high-stack source; M, medium-stack source; L, low-stack (or low-level) source. Also includes coal use by a large steel mill with a high stack.
Sources: Adamson et al. (1996); Janzten (1995).

Table E.3 Manila: Quantity and quality of fuel use, by economic sector

Fuel	*Modern power plants (H)*	*Suboptimal plants/ district heating (M)*	*Large industry (M)*	*Small industry (L)*	*Residential (L)*	*Land transport (L)*
Coal						
Thousands of tons						
Ash content (percent)						
Sulfur content (percent)						
Petroleum products						
Fuel oil						
Thousands of tons	1,198.5		2,405.5	1,037.0	510.0	
Sulfur content (percent)	3.0		1.5	0.5	0.5	
Motor diesel oil						
Thousands of tons						
Sulfur content (percent)						0.5
Gasoline						
Thousands of tons						719.0
Fuelwood						
Thousands of tons						

Note: H, high-stack source; M, medium-stack source; L, low-stack (or low-level) source.
Source: World Bank (1997h).

Table E.4 Mumbai: Quantity and quality of fuel use, by economic sector

Fuel	*Modern power plants (H)*	*Suboptimal plants/ district heating (M)*	*Large industry (M)*	*Small industry (L)*	*Residential (L)*	*Land transport (L)*
Coal						
Thousands of tons	298.0		350.0	250.0	100.0	
Ash content (percent)	12.0		12.0	12.0	12.0	
Sulfur content (percent)	0.5		0.5	0.5	0.5	
Petroleum products						
Fuel oil						
Thousands of tons	927.0		626.0	261.0	480.0	
Sulfur content (percent)	1.0	2.0	1.1	1.3	0.15	
Motor diesel oil						
Thousands of tons						243.4
Sulfur content (percent)						0.5
Gasoline						
Thousands of tons						248.6
Fuelwood						
Thousands of tons				192.0	101.0	

Note: H, high-stack source; M, medium-stack source; L, low-stack (or low-level) source.
Source: World Bank (1997g).

Table E.5 Santiago: Quantity and quality of fuel use, by economic sector

Fuel	*Modern power plants (H)*	*Suboptimal plants/ district heating (M)*	*Large industry (M)*	*Small industry (L)*	*Residential (L)*	*Land transport (L)*
Coal						
Thousands of tons						
Ash content (percent)						
Sulfur content (percent)						
Petroleum products						
Fuel oil						
Thousands of tons		46.0	575.0		476.0	
Sulfur content (percent)		1.0	1.0		0.5	
Motor diesel oil						
Thousands of tons						315.0
Sulfur content (percent)						0.5
Gasoline						
Thousands of tons						740.0
Fuelwood						
Thousands of tons)				327.0	168.0	

Note: H, high-stack source; M, medium-stack source; L, low-stack (or low-level) source.
Sources: World Bank (1994); interim consultant report.

Table E.6 Shanghai: Quantity and quality of fuel use, by economic sector

Fuel	*Modern power plants (H)*	*Suboptimal plants/ district heating (M)*	*Large industry (M)*	*Small industry (L)*	*Residential (L)*	*Land transport (L)*
Coal						
Thousands of tons	11,000.0	3,200.0	11,000.0	2,200.0	1,500.0	
Ash content (percent)	17.0	20.0	20.0	20.0	5.0	
Sulfur content (percent)	1.0	1.0	1.0	1.0	1.0	
Petroleum products						
Fuel oil						
Thousands of tons	369.0		2,800.0	740.0	460.0	
Sulfur content (percent)	0.5		0.5	0.5	0.5	
Motor diesel oil						
Thousands of tons						455.0
Sulfur content (percent)						1.5
Gasoline						
Thousands of tons						840.0
Fuelwood						
Thousands of tons						

Note: H, high-stack source; M, medium-stack source; L, low-stack (or low-level) source.
Sources: World Bank (1997d); World Bank staff (for diesel and gasoline sales).

Notes

1. PM_{10} is particulate matter less than 10 microns in aerodynamic diameter.

2. The use of diesel by railroads and intercity transport is not included in this assessment.

3. These are health impacts attributable to exposure to outdoor (ambient) air pollution only. Exposure to high levels of indoor air pollution in households using solid fuels (coal, wood, and agricultural waste) is not assessed in the study. On a global scale, health impacts from indoor exposure are very significant and are considered greater than those from outdoor pollution (World Bank 1992, 1993; Smith 1993, 1998; WHO 1997). Most of the people who suffer from high levels of indoor pollution are rural, but poor urban families are also affected. The health costs of fuel use would be considerably greater if indoor pollution were taken into account.

4. The category "Power plants" in Figure 1.4 and throughout the study includes only modern, well-controlled power plants with high stacks. This restriction explains the very low contribution of the power sector to local damages. Suboptimal power stations and generators, which are common in Shanghai, as well as Krakow's large district heating boilers, are included in the category "Large boilers." Long-range pollution from power plants and other large sources that are located outside urban agglomerations is not assessed because the assessment is based on fuel use inventory within a city or agglomeration. In the case of Krakow, which is adjacent to a large coal-mining and industrial region, long-range pollution is estimated to increase local damage from large sources by 40–50 percent (authors' estimates based on Adamson et al. 1996). This impact, although significant, does not change the fundamental importance of small sources in local pollution; in Krakow, it would increase the contribution of large sources to local damage from 10 to 14 percent.

5. The fuel prices shown in Figure 1.8 and Table 1.6 are the following 1993 spot market (producer) prices: coal, Australian export; fuel oil, diesel (gas oil), and gasoline prices, Rotterdam product prices. (The gasoline price is for regular unleaded and the fuel oil price is for fuel oil with 1 percent sulfur content, which is close to the average sulfur content of the fuel oil aggregate in the sample.) The local coal prices shown in Figure 1.9 and Table 1.7 are wholesale prices for power plants and large boilers and retail prices for small users. Local prices are for different years within the range 1991–94. The sources are IEA and OECD (1995, 1996); Adamson et al. (1996); Kubota (1996); World Bank 1997f; World Bank staff.

6. For example, emissions factors for diesel vehicles assume that a large share of high-

sulfur diesel is used in uncontrolled or poorly maintained old vehicles that emit substantial amounts of particulates and SO2. The assumptions and emissions factors are modified for each city on the basis of city-specific or country-specific information, including the composition of the vehicle fleet by age and type, where available.

7. USEPA Website <http://www.epa.gov>.

8. A lower level of exposure was assumed for Shanghai because the impact of fuel is assessed for all of Shanghai province (population 13.5 million). The province has a lower overall population density than that of other cities, where the assessment pertains to the area within the immediate city or agglomeration boundaries.

9. In the sample of six cities, fuel combustion contributes 42 percent of total exposure to PM_{10}. Fuel use inventories may be incomplete for some cities (especially for Bangkok and Santiago) because of the limitations of the secondary data available. Thus, the contribution of fuel burning to PM_{10} exposure might be larger. Since major fuel uses are incorporated into our analysis, however, any possible difference is unlikely to exceed a margin of 20 percent for Bangkok and Santiago and is even smaller for other cities.

10. The pre-1997 WHO guidelines for annual average levels of PM_{10} are 40–60 $\mu g/m^3$. In 1997 WHO discontinued its threshold level guidelines for particulates after substantial evidence was accumulated that adverse health effects occurred even at much lower levels of exposure. Organisation for Economic Cooperation and Development (OECD) standards for PM_{10} typically range between 40 and 50 $\mu g/m^3$ (annual average), and the United States and the European Union are currently considering the adoption of stricter standards.

11. Some recent studies (Oberdorster et al. 1995; Seaton et al. 1995; Peters et al. 1997) go even further, indicating that *ultrafine particles* in ambient air may be responsible for the observed health effects because of their high biological and toxicological reactivity. Ultrafine particles are the smallest fraction of fine particulates (typically smaller than 0.05 mm) that exist in a nucleation mode. The most prevalent ambient ultrafine particles are elemental and organic carbon particles (Hilderman et al. 1994).

12. In addition, a study by Zejda et al. (1997) of air pollution and daily mortality in Katowice, Poland, yielded a central estimate of a 0.7 percent change in total mortality per 10 $\mu g/m^3$ change in PM_{10} levels. This result is consistent with Table 3.3. The study is not included in the meta-analysis because a complete report is not available in English.

13. The adverse health impact of lead, in combination with the availability of low-cost options for introducing unleaded gasoline, has hastened a worldwide trend toward phasing out leaded gasoline. In recent years transport in Bangkok, Shanghai, and Mumbai has become lead free, and Santiago is following this path.

14. Shares in the total population of the cities of children under 14 and asthmatics, which are used in some dose-response functions in Table 3.4, were assumed to be 0.27 and 0.05 for each city, on the basis of U.S. data. If city-specific values for these parameters are available, they should be used to obtain more accurate results. However, possible local variations in these values make little difference to the total social costs of health

impacts and do not affect any conclusions from the analysis (see Chapter 6).

15. Monte Carlo (random) simulations were used to generate the distribution of willingness to pay to avoid a case of pollution-related bronchitis, drawing from the distributions for the three variables: willingness to pay to avoid a severe case of chronic bronchitis: the severity level of an average pollution-related case of chronic bronchitis; and the WTP elasticity.

16. Cropper and Krupnick (1990) estimate that the average cost of a hospital stay for respiratory disease is US$1,801 (in 1977 dollars). This corresponds to US$5,785 in 1990 prices, using the consumer price index (CPI) for medical care. Given an average duration of a hospital stay of about 9.5 days, weekly earnings of US$421, and a five-day week, the cost of lost output would be US$85 per day. The COI for an RHA is therefore US$6,589. Rowe et al. (1986) assess the medical cost of an ERV as US$90 in 1986 prices. Using the medical care CPI, this would be US$135 in 1990 dollars, and adding the cost of a day's wages brings the cost to US$220.

17. Note that the impact of ozone is not considered; this tends to lower values for NO_x.

References

Abbey, D. E., P. K. Mills, F. F. Peterson, and W. L. Beeson. 1991. "Long-Term Ambient Concentrations of Total Suspended Particulates and Oxidants as Related to Incidence of Chronic Disease in California Seventh-Day Adventists." *Environmental Health Perspectives* 94: 43–50.

———. 1993. "Long-Term Ambient Concentrations of Total Suspended Particulates, Ozone and Sulfur Dioxide and Respiratory Symptoms in a Non-Smoking Population." *Archives of Environmental Health* 48 (1): 33–46.

Adamson, Seabron, Robin Bates, Robert Laslett, and Alberto Pototschnig. 1996. *Energy Use, Air Pollution, and Environmental Policy in Krakow: Can Economic Incentives Really Help?* World Bank Technical Paper 308, Energy Series. Washington, D.C.

Alberini, A., et al. 1997. "Valuing Health Effects of Air Pollution in Developing Countries: The Case of Taiwan." *Journal of Environmental Economics and Management* 34: 107–26.

Alberini, A., and A. Krupnick. 1998. "Air Quality and Episodes of Acute Respiratory Illness in Taiwan Cities: Evidence from Survey Data." *Journal of Urban Economics* 44: 68–92.

Anand, Sudhir, and Kara Hanson. 1997. "Disability-Adjusted Life Years: A Critical Review." *Journal of Health Economics* 16: 685–702.

ApSimon, H., and D. Cowell. 1996. "The Benefits of Reduced Damage to Buildings from Abatement of Sulphur Dioxide Emissions." *Energy Policy* 24 (7): 651–54.

ASEP (Atmospheric Science Expert Panel). 1997. "Health and Environmental Impact Assessment Panel Report, Joint Industry/Government Study." Environment Canada. Hull, Quebec, Canada.

Azar, C. 1999. "Weight Factors in Cost-Benefit Analysis of Climate Change." *Environmental and Resource Economics* 13: 249–68.

Azar, C., and T. Sterner. 1996. "Discounting and Distributional Considerations in the Context of Global Warming." *Ecological Economics* 19: 169–84.

Balson, W., R. T. Carson, and R. C. Mitchell. 1990. *Development and Design of a Contingent Valuation Survey for Measuring the Public's Value for Visibility Improvements in the Grand Canyon National Park.* Los Altos, Calif.: Decision Focus Inc.

Biddle, J., and G. Zarkin. 1988. "Worker Preferences and Market Compensation for Job Risk." *Review of Economic Statistics* 70 (4): 660–66.

Booz, et al. 1970. *Study to Determine Residential Soiling Cost of Particulate Air Pollution.* U.S. Department of Health, Education and Welfare, Environmental Health Service, National Air Pollution Control Administration, Raleigh, N.C.

Borja-Aburto, V., D. P. Loomis, S. I. Bangdiwala, C. M. Shy, and R. A. Rascon-Pacheco. 1997. "Ozone, Suspended Particulates, and Daily Mortality in Mexico City." *American Journal of Epidemiology* 145 (3): 258–68.

Brajer, V., et al. 1991. "The Value of Cleaner Air: An Integrated Approach." *Contemporary Policy Issues* 9 (4): 81–91.

Brookshire, D., R. C. D'Arge, W. D. Shulze, and M. A. Thayer. 1979. *Methods Development for Assessing Air Pollution Control Benefits. Vol. 2: Experiments in Valuing Non-Market Goods. A Case Study of Alternative Benefit Measures of Air Pollution Control in the South Coast Air Basin of Southern California.* Washington, D.C.: U.S. Environmental Protection Agency.

Burtaw, D., A. Krupnick, E. Mansur, D. Austin, and D. Farrell. 1997. *The Costs and Benefits of Reducing Acid Rain.* Discussion Paper. Resources for the Future, Washington, D.C.

Calthrop, E. 1996. *Methodologies for Calculating the Damage to Buildings and Materials.* London: FTCE Ltd.

Chestnut, L., and D. Violette. 1984. *Estimates of Willingness to Pay for Pollution-Induced Changes in Morbidity.* Available from National Technical Information Service, Springfield, Va.

Chestnut, L., B. Ostro, and N. Vichit-Vadakan. 1997. "Transferability of Air Pollution Control Health Benefits Estimates from the United States to Developing Countries: Evidence from the Bangkok Study." *American Journal of Agricultural Economics* 79 (5): 1630–35.

Cline, W. 1992. *The Economics of Global Warming.* Washington, D.C.: Institute for International Economics.

———. 1993. *The Economics of Climate Change.* Washington, D.C.: Institute for International Economics.

Cowell, D., and H. ApSimon. 1996. "Estimating the Cost of Damage to Buildings by Acidifying Atmospheric Pollution in Europe." *Atmospheric Environment* 30 (17): 2959–68.

Cropper, Maureen. 1981. "Measuring the Benefits from Reduced Morbidity." *American Economic Review* 71 (2): 235–40.

Cropper, Maureen, and Alan J. Krupnick. 1990. "Social Costs of Chronic Heart and Lung Disease." Resources for the Future Discussion Paper QE 89-16-REV. Washington, D.C.

Cropper, Maureen, and Nathalie Simon. 1996. "Valuing the Health Effects of Air Pollution." DEC Note 7. World Bank, Washington, D.C.

Cropper, Maureen, and F. Sussman. 1990. "Valuing Future Risks to Life." *Journal of Environmental Economics and Management* 20 (2): 160–74.

Cropper, Maureen L., Nathalie B. Simon, Anna Alberini, and P. K. Sharma. 1997. "The Health Effects of Air Pollution in Delhi, India." Policy Research Working Paper 1860. Development Economics Research Group, World Bank, Washington, D.C.

Day, B. 1999. "A Meta-Analysis of Wage Risk Estimates of the Value of Statistical Life." Centre for Social and Economic Research on the Global Environment, University College London. Processed

Dennis, R. 1978. *The Smeared Concentration Approximation Method: A Simplified Air Pollution Dispersion Methodology for Regional Analysis.* Laxenburg, Austria: International Institute for Applied Systems Analysis.

Desvousges, W., F. Johnson, and H. Banzhaf. 1995. *Assessing Environmental Externality Costs for Electricity Generation.* Vol. 5. Durham, N.C.: Triangle Economic Research.

Dockery, Douglas W., C. A. Pope, X. Xiping, J. Spengler, J. Ware, M. Fay, B. Ferris, and F. Speizer. 1993. "An Association between Air Pollution and Mortality in Six Cities." *New England Journal of Medicine* 329 (24): 1753–59.

Downing, Robert, Ramesh Ramankutty, and Jitendra Shah. 1997. *RAINS-Asia: An Assessment Model for Acid Deposition in Asia.* Directions in Development series. Washington, D.C.: World Bank.

EC (European Commission), Directorate-General XII. 1995. *ExternE Externalities of Energy.* Brussels.

ECOTEC Ltd. 1994. *An Evaluation of the Benefits of Reduced Sulphur Dioxide Emissions.* Report to the U.K. Department of the Environment. Birmingham, U.K.

Eskeland, Gunnar S., and Shantayanan Devarajan. 1995. *Taxing Bads by Taxing Goods: Pollution Control with Presumptive Charges.* Directions in Development series. Washington, D.C.: World Bank.

Eskeland Gunnar S., and Jian Xie. 1998. "Acting Globally While Thinking Locally: Is the Global Environment Protected by Transport Emission Control Programs?" Policy Research Working Paper 1975. Public Economics, Development Research Group, and Global Environment Unit, Environment Department, World Bank, Washington, D.C.

Evans, J., T. Tosteson, and P. L. Kinney. 1984. "Cross-Sectional Mortality Studies and Air Pollution Risk Assessment." *Environment International* 10: 55–83.

Eyre, N., T. Downing, R. Hoekstra, K. Rennings, and R. Tol. 1997. *Global Warming Damages.* Final Report of the ExternE Global Warming Sub-Task, DGXII. Brussels: European Commission.

Fairley, D. 1990. "The Relationship of Daily Mortality to Suspended Particulates in Santa Clara County 1980–86." *Environmental Health Perspectives* 89: 159–68.

Fankhauser, Samuel. 1994. "The Social Costs of Greenhouse Gas Emissions: An Expected Value Approach." *Energy Journal* 15 (2): 157–84.

———.1995. *Valuing Climate Change: The Economics of the Greenhouse.* London: Earthscan.

Fankhauser, Samuel, Richard S. J. Tol, and David W. Pearce. 1997. "The Aggregation of Climate Change Damages: A Welfare Theoretic Approach." *Environment and Resource Economics* 10 (3, October): 249–66.

———. 1998. "Extensions and Alternatives to Climate Change Impact Valuation: On the Critique of IPCC Working Group III's Impact Estimates." *Environment and Development Economics* 3 (February): 59–81.

Feliu S., M. Morcillo, and J. Feliu. 1993. "The Prediction of Atmospheric Corrosion from Meteorological and Pollution Parameters: I (Annual Corrosion) and II (Long-Term Forecast)." *Corrosion Science* 34 (3): 403–22.

Fisher, A., L. Chestnut, and D. Violette. 1989. "The Value of Reducing Risks of Death: A Note on New Evidence." *Journal of Policy Analysis and Management* 8 (1): 88–100.

Glomsrød, S., O. Godal, J. Fr. Henriksen, S. E. Haagenrud, and T. Skancke. 1996. "Air Pollution Impacts and Values: Corrosion Costs of Building Materials and Cars in Norway." Oslo: Norwegian Pollution Control Authority.

Haagenrud, S., and J. Henriksen. 1996. "Survey of Dose-Response Functions for Corrosion Damage on Materials." Paper read to United Nations Economic Commission for Europe (UNECE) Workshop on Economic Evaluation of Air Pollution Abatement and Damage to Buildings, Including Cultural Heritage, Stockholm.

Hilderman, L. M., D. Klinedinst, G. Klouda, L. Currie, and G. Case. 1994. "Sources of Urban Contemporary Carbon Aerosol." *Environmental Science and Technology* 28 (9): 1585–76.

Hohmeyer, Olav, and Richard L. Ottinger. 1991. *External Environmental Costs of Electric Power: Analysis and Internalization.* Heidelberg, Germany: Springer-Verlag.

Holgate, Stephen T., Jonathan M. Samet, Hillel S. Koren, and Robert L. Maynard. 1999. *Air Pollution and Health.* San Diego, Calif.: Academic Press.

IEA (International Energy Agency) and OECD (Organisation for Economic Co-operation and Development). 1991. *Greenhouse Gas Emissions: The Energy Dimension.* Paris.

———. 1995. *Coal Trade Statistics.* Paris: OECD.

———. 1996. *Monthly Oil Market Report* (December). Paris: OECD.

IEI (Industrial Economics Incorporated). 1992. "Revisions to the Proposed Value of Life." Memo. Cambridge, Mass.

Ito, K., and G. Thurston. 1996. "Daily PM10/ Mortality Associations: An Investigation of At Risk Subpopulations." *Journal of Exposure Analysis and Environmental Epidemiology* 6: 79–95.

Jantzen J. 1995. "Urban Transport and Air Pollution: Trends and Policy Options." Interim report for the World Bank. Washington, D.C.

Johannesson, M., and P. Johansson. 1996. "To Be, or Not to Be, That Is the Question: An Empirical Study of the WTP for an Increase in Life Expectancy at an Advanced Age." *Journal of Risk and Uncertainty* 13: 136–74.

Jones-Lee, M., M. Hammerton, and P. Philips. 1985. "The Value of Safety: The Results of a National Sample Survey." *Economic Journal* 95 (377): 49–72.

Kaplan, R., et al. 1993. "The Quality of Well-Being Scale: Rationale for a Single Quality of Life Index." In S. Walker and R. Rosser, eds., *Quality of Life Assessment: Key Issues in the 1990s.* Dordrecht, the Netherlands: Kluwer.

Katsouyanni, K., G. Touloumi, C. Spix, J. Schwartz, F. Balducci, S. Medina, G. Rossi, B. Wojtyniak, J. Sunyer, L. Bacharova, J. P. Schouten, A. Ponka, and H. R. Anderson. 1997. "Short Term Effects of Ambient Sulphur Dioxide and Particulate Matter on Mortality in 12 European Cities: Results from Time Series Data from the APHEA Project." *British Medical Journal* (314): 1658–63.

Kinney P., I. Kazuhiko, and G. Thurston. 1995. "A Sensitivity Analysis of Mortality/PM10 Associations in Los Angeles." *Inhalation Toxicology* 7: 59–69.

Krupnick Alan J., and D. Burtaw. 1996. "The Social Costs of Electricity: Do the Numbers Add Up?" *Resource and Energy Economics* 18: 423–66.

Krupnick Alan J., and Maureen Cropper. 1992. "The Effect of Information on Health Risk Valuations." *Journal of Risk and Uncertainty* 5: 29–48.

Krupnick, Alan J., and Paul R. Portney. 1991. "Controlling Urban Air Pollution: A Benefit-Cost Assessment." *Science* 252: 522–28.

Krupnick Alan J., Anna Alberini, Maureen Cropper, and Nathalie Simon, with Kenshi Itaoka and Mokoto Akai. 1999. "Mortality Risk Valuation for Environmental Policy." Discussion Paper 99-47. Resources for the Future, Washington, D.C.

Kubota, S. 1996. *Natural Gas Trade in Asia and the Middle East.* IEN Occasional Paper 8. Washington, D.C.

Kucera, V. 1996. "Effects of Nitrogen Pollutants and Ozone on Damage to Materials." Paper read to United Nations Economic Commission for Europe (UNECE) Workshop on Economic Evaluation of Air Pollution Abatement and Damage to Buildings, Including Cultural Heritage, Stockholm.

Kucera, V., J. Henriksen, J. Knotkova, and C. Sjostrom. 1993. *Models for the Calculation of Corrosion Cost Caused by Air Pollution and Its Application in Three Cities.* Copenhagen: Nordic Council of Ministers.

Kucera, V., et al. 1995. *Statistical Analysis of 4-Year Materials Exposure and Acceptable Deterioration and Pollution Levels.* Report 18. UNECE ICP on Effects on Materials Including Historic and Cultural Monuments. New York.

Landrieu, G. 1997. "Visibility Impairments by Secondary Ammonium, Sulphates, Nitrates and Organic Particles." Draft note prepared for the United Nations Economic Commission for Europe (UNECE) Convention on LRTAP Task Force on the Economic Aspects of Abatement Policies, Copenhagen.

Lave, L., and E. Seskin. 1977. *Air Pollution and Human Health.* Baltimore, Md.: John Hopkins University Press.

Lee, R., A. J. Krupnick, and D. Burtraw. 1995. *Estimating Externalities of Electric Fuel Cycles: Analytical Methods and Issues, Estimating Externalities of Coal Fuel Cycles.* Washington, D.C.: McGraw-Hill/Utility Data Institute. (See also additional volumes for other fuel cycles.)

Lipfert, Frederick W. 1989. "Atmospheric Damage to Calcareous Stones: Comparison and Reconciliation of Recent Experimental Findings." *Atmospheric Environment* 23: 415–29.

———.1994. *Air Pollution and Community Health: A Critical Review and Data Sourcebook.* New York: Van Nostrand Reinhold.

Loehman, E., and V. H. De. 1982. "Application of Stochastic Choice Modeling to Policy Analysis of Public Goods: A Case Study of Air Quality Improvements." *Review of Economics and Statistics* 64 (3): 474–80.

Loehman, E., et al. 1980. *Measuring the Benefits of Air Quality Improvements in the San Francisco Bay Area. Part I: Study Design and Property Value Study.* Menlo Park, Calif.: SRI International.

Lovei, Magda. 1998. *Phasing Out Leaded Gasoline: Worldwide Experience and Policy Implications.* World Bank Technical Paper 397. Washington, D.C.

Maddison, David. 1994. "The Shadow Price of Greenhouse Gases and Aerosols." Centre for Social and Economic Research on the Global Environment, University College London and University of East Anglia. Processed.

———. 1997. "The Amenity Value of Climate: The Household Production Function Approach." Paper presented to the European Association of Environmental and Research Economists conference, Tilburg, the Netherlands, June 26–28.

Markandya, Anil. 1991. "Measuring the External Costs of Fuel Cycles." In Olav Hohmeyer and Richard L. Ottinger, eds., *External Environmental Costs of Electric Power: Analysis and Internalization.* Heidelberg, Germany: Springer-Verlag.

MATHTEC. 1984. *Economic Benefits of Reduced Acidic Deposition on Common Building Materials:* Methods Assessment. Princeton, N.J.

McClelland, G., W. Schulze, D. Waldman, J. Irwin, T. Stewert, L. Deck, and M. Thayer. 1991. *Valuing Eastern Visibility: A Field Test of the Contingent Valuation Method.* Washington, D.C.: U.S. Environmental Protection Agency.

Mendelsohn, Robert, and Ariel Dinar. 1999. "Climate Change, Agriculture, and Developing Countries: Does Adaptation Matter?" *World Bank Research Observer* 14 (2): 277–305.

Mendelsohn, Robert, and James E. Neumann, eds. 1999. *The Impact of Climate Change on the United States Economy.* New York: Cambridge University Press.

Mendelsohn, Robert, W. Morrison, M. Schlesinger, and N. Andronova. 1996. *Global Impact Model for Climate Change.* School of Forestry, Yale University, New Haven, Conn. Processed.

Miller, T. 1990. "The Plausible Range for the Value of Life: Red Herrings among the Mackerel." *Journal of Forensic Economics* 3 (3): 17–37.

Moore, M., and K. Viscusi. 1988. "The Quantity Adjusted Value of Life." *Economic Inquiry* 26 (3): 369–88.

MRI. 1980. *Damage Functions for Air Pollutants.* Washington, D.C.: U.S. Environmental Protection Agency.

Murray Christopher J. L., and Alan D. Lopez, eds. 1996. *The Global Burden of Disease. A Comprehensive Assessment of Mortality and Disability from Diseases, Injuries, and Risk Factors in 1990 and Projected to 2020.*

Global Burden of Disease and Injury Series, no. 1. Cambridge, Mass.: Harvard School of Public Health. Distributed by Harvard University Press.

Noll, K., et al. 1968. "Visibility and Aerosol Concentration in Urban Air." *Atmospheric Environment* 2: 465–75.

Nordhaus, William D. 1991. "To Slow or Not to Slow: The Economics of the Greenhouse Effect." *Economic Journal* 101 (407): 920–37.

———. 1994. *Managing the Global Commons: The Economics of Climate Change.* Cambridge, Mass.: MIT Press.

———. 1998. *New Estimates of the Economic Impacts of Climate Change.* New Haven, Conn.: Yale University Press.

Nordhaus, William D., and J. Boyer. Forthcoming. *Rolling the DICE Again: The Economics of Global Warming.* Cambridge, Mass.: MIT Press.

Oberdorster, G., R. Gelein, J. Ferin, and B. Weiss. 1995. "Association of Particulates Air Pollution and Acute Mortality: Involvement of Ultrafine Particles?" *Inhalation Toxicology* 7: 111–24.

OECD (Organisation for Economic Co-operation and Development). 1996. "Implementation Strategies for Environmental Taxes." Paris.

Ostro, Bart. 1994. "Estimating the Health Effects of Air Pollution: A Methodology with Application to Jakarta." Policy Research Working Paper 1301. Policy Research Department, World Bank, Washington, D.C.

———. 1996. *A Methodology for Estimating Air Pollution Health Effects.* WHO/EHG/96.5. Geneva: World Health Organization.

Ostro, Bart, José Miguel Sanchez, Carlos Aranda, and Gunnar S. Eskeland. 1995. "Air Pollution and Mortality: Results from Santiago, Chile." Policy Research Working Paper 1453. Policy Research Department, World Bank, Washington, D.C.

Ostro, Bart, L. Chestnut, N. Vichit-Vadakan, and A. Laixuthai. 1998. "The Impact of Fine Particulate Matter on Mortality in Bangkok, Thailand." Paper presented to the Air and Waste Management Association Specialty Conference on Fine Particles. Air and Waste Management Association, Pittsburgh, Pa.

Ottinger Richard L., et al. 1991. *Environmental Costs of Electricity.* Pace University Center for Environmental Legal Studies. White Plains, N.Y.: Oceana Press.

Ozkaynak, H., and G. Thurston. 1987. "Associations between 1980 US Mortality Rates and Alternative Measures of Airborne Particle Concentrations." *Risk Analysis* 7: 449–61.

Pearce, David. 1997. "Damage to Buildings and Materials." Centre for Social and Economic Research on the Global Environment, University College London and University of East Anglia. Processed.

———. 2000. "A Note on Climate Change Costs." University College London. Processed.

Pearce David, C. Bann, and S. Georgiou. 1992. *The Social Costs of Fuel Cycle: A Report to the UK Department of Energy* Centre for Economic and Social Research on the Global Environment. University College London and University of East Anglia,

Pearce, David, W. R. Cline, A. Achanta, S. Fankhauser, R. Pachauri, R. Tol, and P. Vellinga. 1996. "The Social Costs of Climate Change: Greenhouse Damage and the Benefits of Control." In Intergovernmental Panel on Climate Change, *Climate Change 1995: Economic and Social Dimensions of Climate Change,* 183–224. Cambridge, U.K.: Cambridge University Press.

Peck, Stephen C., and Thomas J. Teisberg. 1992. "CETA: A Model for Carbon Emissions Trajectory Assessment." *Energy Journal* 13 (1): 55–77. Cited in Pope, et al. 1995; WHO 1995.

Peters, A., H. E. Wichmann, T. Tuch, J. Heinrich, and J. Heyder. 1997. "Respiratory Effects

Are Associated with the Number of Ultrafine Particles." *American Journal of Respiratory Critical Care Medicine* 155 (4, April): 1376–83.

Pope, C. Arden, and D. W. Dockery. 1994. "Acute Respiratory Effects of Particulate Air Pollution." *Annual Review of Public Health* 15: 107–32.

Pope, C. A., J. Schwartz, and M. R. Ransom. 1992. "Daily Mortality and PM10 Pollution in Utah Valley." *Archives of Environmental Health* 47: 211–17.

Pope, C.A., M. J. Thun, M. M. Namboodiri, D. W. Dockery, J. S. Evans, F. E. Speizer, and C. W. Heath, Jr. 1995. "Particulate Air Pollution as a Predictor of Mortality in a Prospective Study of U.S. Adults." *American Journal of Respiratory Critical Care Medicine,* no. 151: 669–74.

Radian International. 1997. *PM Abatement Strategy for the Bangkok Metropolitan Area.*

Rae, D. 1983. *Benefits of Improving Visual Air Quality in Cincinnati: Results of a Contingent Valuation Survey.* Palo Alto, Calif.: EPRI.

Repetto, R. 1981. "The Economics of Visibility Protection: On a Clear Day You Can See a Policy." *Natural Resources Journal* 21 (2): 355–70.

Ridker, R. 1967. *Economic Costs of Air Pollution.* New York: Praeger.

Roughgarden, T., and S. Schneider. 1999. "Climate Change Policy: Quantifying Uncertainties for Damages and Optimal Carbon Taxes." *Energy Policy* 27: 415–29.

Rowe, R., and V. Chestnut. 1985. *Oxidants and Asthmatics in Los Angeles: A Benefits Analysis.* Washington, D.C.: U.S. Environmental Protection Agency.

Rowe, R., et al. 1986. *The Benefits of Air Pollution Control in California.* Boulder, Colo.: Energy and Resource Consultants Inc.

Rowe, R., C. Lang, L. Chestnut, D. Latimer, D. Rae, S. Bernow and D. White. 1995. *The New York Electricity Externalities Study,* vols. 1 and 2. New York: Oceana Publications.

Saldiva, Paulo, et al. 1995. "Air Pollution and Mortality in Elderly People: Time-Series Study in São Paulo, Brazil." *Archives of Environmental Health* 50 (2): 159–63.

Schulze, W., et al. 1983. "The Economic Benefits of Preserving Visibility in the National Parklands of the South-West." *National Resources Journal* 23: 149–73.

Schwartz, Joel. 1993. "Air Pollution and Daily Mortality in Birmingham, Alabama." *American Journal of Epidemiology* 137: 1136–47.

———. 1994a. "Air Pollution and Daily Mortality: A Review and Meta Analysis." *Environmental Research,* no. 64: 36–52.

———. 1994b. "Societal Benefits of Reducing Lead Exposure." *Environmental Research,* no. 66: 105–24.

Schwarz, Joel, and Douglas W. Dockery. 1992a. "Increased Mortality in Philadelphia Associated with Daily Air Pollution Concentrations." *American Review of Respiratory Disease* 145: 600–604.

———. 1992b. "Particulate Air Pollution and Daily Mortality in Steubenville, Ohio." *American Journal of Epidemiology* 135: 12–20.

Schwartz, J, C. Spix, G. Touloumi, L. Bacharova, T. Barumamdzadeh, A. le Tertre, T. Piekarksi, A. Ponce de Leon, A. Ponka, G. Rossi, M. Saez, and J. P. Schouten. 1996. "Methodological Issues in Studies of Air Pollution and Daily Counts of Deaths or Hospital Admissions." *Journal of Epidemiology and Community Health* 50: S3–S11.

Seaton, A., W. MacNee, K. Donaldson, and D. Godden. 1995. "Particulate Air Pollution and Acute Health Effects." *Lancet* 345: 176–78.

Sebastian, Iona, Kseniya Lvovsky, and Henk de Koning. 1999. "Decision Support Systems for Integrated Pollution Control." World Bank, Washington, D.C.

Simon, Nathalie B., Maureen L. Cropper, Anna Alberini, and Seema Arora. 1999. "Valuing Mortality Reductions in India: A Study of Compensating Wage Differentials." Policy Research Working Paper 2078. Infrastructure and Environment, Development Research Group, World Bank, Washington, D.C.

Smith, K. R. 1993. "Fuel Combustion, Air Pollution Exposure, and Health: The Situation in Developing Countries." *Annual Review of Energy and Environment* 18: 529–66.

———. 1998. *Indoor Air Pollution in India: National Health Impacts and the Cost-Effectiveness of Intervention*. Mumbai: Indira Gandhi Institute for Development Research.

Streets, D., L. Carter, L. Hedayat, G. Carmichael, and R. Arndt. 1997. "The Potential for Advanced Technology to Improve Air Quality and Human Health in Shanghai." Submitted to *Environmental Management*, December 3, 1997.

Styer, P., N. McMillan, F. Gao, J. Davis, and J. Sacks. 1995. "Effect of Outdoor Airborne Particulate Matter on Daily Death Counts." *Environmental Health Perspectives* 103: 490–97.

Summers, R., and A. Heston. 1995. "A Note on Estimates of GDP per Capita Based on Exchange Rate and PPP Conversions: How Well Does Each Explain Things?" Draft. World Bank, Washington, D.C.

Sunyer, J., J. Anto, J. Castellsague, M. Saez, and A. Tobias. 1996. "Air Pollution and Mortality in Barcelona." *Journal of Epidemiology and Community Health* 50: S77–S80.

TER (Triangle Economic Research). 1995. *Assessing Environmental Externality Costs for Electricity Generation*. Prepared for Northern States Power Company, Minnesota. Durham, N.C.

———. 1996. Valuing Morbidity: An Integration of the Willingness to Pay and Health Status Index Literatures. Durham, N.C.

Titus, J. 1992. "The Costs of Climate Change to the United States." In S. Majumdar, ed., *Global Climate Change: Implications, Challenges and Mitigation Measures*. Easton, Pa.: Pennsylvania Academy of Science.

Tol, R. 1995. "The Damage Costs of Climate Change: Towards More Comprehensive Estimates." *Environmental and Resource Economics* 5: 353–74.

———. 1999. "The Marginal Costs of Greenhouse Gas Emissions." *Energy Journal* 20 (1): 61–81.

Tol, R., S. Fankhauser, and D. W. Pearce. 1996. "Equity and the Aggregation of the Damage Costs of Climate Change." In V. Nacicenovic, W. Nordhaus, R. Richels, and F. Toth, eds., *Climate Change: Integrating Science, Economics and Policy*, 167–78. Laxenburg, Austria: International Institute for Applied Systems Analysis.

——— 1999. "Empirical and Ethical Arguments in Climate Change Impact Valuation and Aggregation." In F. Toth, ed., *Fair Weather? Equity Concerns in Climate Change*, 65–79. London: Earthscan.

Tolley, G., A. Randall, A. Frankel, and R. Fabian. 1984. *Establishing and Valuing the Effects of Improved Visibility in the Eastern United States*. Washington, D.C.: U.S. Environmental Protection Agency.

Tolley, G. S., L. Babcock, M. Berger, A. Bilotti, G. Blomquist, R. Fabian, G. Fishelson, C. Kahn, A. Kelly, D. Kenkel, R. Kumm, T. Miller, R. Ohsfeldt, S. Rosen, W. Webb, W. Wilson, and M. Zelder. 1986. "Valuation of Reductions in Human Health Symptoms and Risk." In *Contingent Valuation Study of Light Symptoms and Angina*. Washington, D.C.: U.S. Environmental Protection Agency.

United Kingdom, Department of the Environment. 1994. *Air Pollution in the UK 1992/3.* London: Her Majesty's Stationery Office.

UNCHS (United Nations Centre for Human Settlements). 1986. *Global Report on Human Settlements.* Oxford, U.K.: Oxford University Press.

———. 1996. *Global Report on Human Settlements.* Oxford, U.K.: Oxford University Press.

U.S. Bureau of the Census. 1996. *Statistical Abstract of the United States: 1996.* Washington, D.C.

USEPA (U.S. Environmental Protection Agency). 1986. *Compilation of Air Pollutant Emission Factors,* Supplement A. USA. Washington, D.C.

———.1997. *The Costs and Benefits of the Clean Air Act.* Washington, D.C.

Viscusi, W. Kip. 1992. *Fatal Trade-Offs: Public and Private Responsibilities for Risk.* New York: Oxford University Press.

———. 1993. "The Value of Risks to Life and Health." *Journal of Economic Literature* 31: 1912–46.

Viscusi, W. Kip, and W. Evans. 1990. "Utility Functions That Depend on Health Status: Estimates and Economic Implications." *American Economic Review* 80 (3): 353–74.

Viscusi, W. Kip, W. Magat, and J. Huber. 1991. "Pricing Health Risks: Survey Assessments of Risk-Risk and Dollar-Risk Tradeoffs." *Journal of Environmental Economics and Management* 21 (1): 32–51.

Watson, W., and J. Jaksch. 1982. "Air Pollution: Household Soiling and Consumer Welfare Losses." *Journal of Environmental Economics and Management* 9: 248–62.

WEC (World Energy Council). 1992. *International Energy Data.* London.

WHO (World Health Organization). 1989. *Management and Control of the Environment.* Geneva.

———. 1995. *Update and Revision of Air Quality Guidelines for Europe.* Geneva.

———. 1997. *Health and Environment in Sustainable Development.* Geneva.

WHO (World Health Organization) and UNEP (United Nations Environment Programme). 1992. *Urban Air Pollution in Megacities of the World.* Oxford, U.K.: Basil Blackwell.

World Bank. 1992. *World Development Report 1992: Development and the Environment.* New York: Oxford University Press.

———. 1993. *World Development Report 1993: Investing in Health.* New York: Oxford University Press.

———. 1994. *Chile: Managing Environmental Problems: Economic Analysis of Selected Issues.* Washington, D.C.

———. 1995a. "Benxi Energy and Air Pollution Management Study." Technical Paper. Washington, D.C.

———. 1995b. *World Development Report 1995: Workers in an Integrating World.* New York: Oxford University Press.

———. 1996. *India: Energy Sector: Issues and Options.* Washington, D.C.

———. 1997a. *Can the Environment Wait? Priorities for East Asia.* Washington, D.C.

———. 1997b. "China: Waigaoqiao Power Plant Project." Staff Appraisal Report. Washington, D.C.

———. 1997c. *Clear Water, Blue Skies: China's Environment in the New Century.* Washington, D.C.

———. 1997d. "Evaluation of Emission Strategies for Shanghai, China: Preliminary Results." Washington, D.C. Processed.

———. 1997e. *Kingdom of Morocco: Environment Review.* Washington, D.C.

———. 1997f. "Philippines: Energy Strategy and Pricing Study." Washington, D.C.

———. 1997g. *Urban Air Quality Management in Asia (URBAIR): Greater Mumbai Report.*

World Bank Technical Paper 380. Washington, D.C.

———. 1997h. *Urban Air Quality Management in Asia (URBAIR): Metro Manila Report.* World Bank Technical Paper 381. Washington, D.C.

———. 1998. *World Development Indicators 1998.* Washington, D.C.

WRI (World Resources Institute). 1997. *World Resources.* New York: Oxford University Press.

Xu, X., D. Dockery, J. Gao, and Y. Chen. 1994. "Air Pollution and Daily Mortality in Residential Areas of Beijing, China." *Archives of Environmental Health* 49 (4): 216–22.

Zejda J. E., et al. 1997. "Ocena Zagrozenia Zdrowia w Nastepstwie Zanieczyszczenia Powietrza Atmosferycznego na Obszarze Miejskim Wojewodztwa Katowickiego: Oszacowanie Ryzyka i Opracowanie Schematu Komunikowanie Ryzyka w Odniesieniu do Poziomow Zanieczyszczen Pylowych i Gazowych" [The Evaluation of Health Risks Related to Ambient Air Pollution in the Urban Area of Katowice Voivodship: Risk Assessment Communication Pertinent to Current Levels of Particulate and Gaseous Pollutants]. Institute of Occupational Medicine and Environmental Health, Sosnowiec, Poland.

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