ECONOMIC COSTS OF AIR POLLUTION WITH SPECIAL REFERENCE TO INDIA

KSENIYA LVOVSKY
SOUTH ASIA ENVIRONMENT UNIT
WORLD BANK[1]

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Abstract

Poor air quality is one of the most serious environmental problems in urban areas around the world, especially in developing countries. Recent studies that assess and value the adverse health impacts of exposure to particulates reveal the magnitude of the costs to society that calls for immediate actions. The paper shows that India appears to bear a very high level of these costs by international comparison. It reviews some latest findings in quantifying the impact of exposure to particulates on mortality with a special reference to India, and discusses the issues of economic valuation of sickness and premature death due to air pollution, with the focus on developing countries. Further, the paper analyzes, drawing upon a case study of Mumbai, the relative effects of various pollution sources on the exposure levels and health outcomes, as well as the health benefits of specific control measures and policies. The concluding section highlights a set of issues and recommendations regarding a better integration of environmental health considerations into pollution management decisions.

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[1] The paper essentially relies on the World Bank study, *Environmental Costs of Fossil Fuels*, undertaken by the team comprising K. Lvovsky, D. Maddison. B. Ostro, G. Hughes and D. Pearce. The report of the entire study is available from the South Asia Social Development and Environment Unit of the World Bank. The contributions by Bart Ostro to the discussion of the health effects should be specially acknowledged. The views presented in this paper are entirely those of the author. The World Bank and its member countries are neither responsible for nor do they endorse the analysis and conclusions.

1. The health burden of air pollution in India: an international comparison assessment

Poor air quality is one of the most serious environmental problems in urban areas around the world, especially in developing countries. An assessment of health damages from exposure to the high levels of particulates in 126 cities worldwide where the annual mean levels exceed 50 ug/m3[2] reveals that these damages may amount to near 130,000 premature deaths; over 500,000 new cases of chronic bronchitis and many more lesser health effects each year. In aggregate terms, this is equivalent to 2.8 million DALYs lost for this sample of nearly 300 million people or 9 DALYs lost per 1000 exposed residents. Highly polluted megacities of China and India account for 82 percent of these DALYs, and low- and middle-income countries together account for 98 percent of DALYs lost for this sample. India, which is represented by 12 largest cities in the sample, ranks second after China, bearing 30 percent of the total DALYs lost, or 12 DALYs lost per 1000 residents of those cities. By comparison, exposure to particulates imposes the burden of 2 DALYs lost per 1000 residents in the cities from a high-income group of countries in the sample, and 3 DALYs lost per 1000 urban residents for middle-income countries. When expressed in monetary values and as a share of the respective incomes (GDP/capita), health damages jump from 3 percent for the sample average up to 9 percent for India and 12 percent for China.

In other words, the costs to society, part of which is direct productivity loss, due to air pollution in largest India cities are as high as nearly one-tenth of the income generated in these cities from all economic activities. Notwithstanding all the uncertainties around such estimates[3], this analysis clearly shows that ***India suffers from a disproportionally heavy health burden of urban air pollution by international comparison.***

2. How significant are the health costs in respect to other damages due to air emissions? - A study of six cities

A significant source of urban air pollution is the combustion of fuels by power plants, industrial boilers, residential stoves and vehicle engines. In addition to imposing a heavy burden of ill-health, by-products of fuel burning cause a variety of non-health damages ranging from local impacts on visibility and buildings to long-range acid depositions to greenhouse gas emissions that may change the Earth's climate. On the other hand, fuel use provides vital energy and transportation services, the demand for which should be taken into account when designing policies and measures to combat the adverse environmental effects from the use of fuels.

The World Bank undertook a study that assessed the magnitude of various damages in urban areas that may be attributed to different fuels, sectors and pollutants (see Lvovsky et al., forthcoming). These damages considered in the study include: the adverse health effects of exposure to air pollution in urban areas; local non-health effects, i.e. reduction in visibility, soiling and material damages; and global climate change impacts. The analysis was applied to six large cities in different parts of the world suffering from the high levels of air pollution -- Bangkok, Thailand; Krakow, Poland; Manila, Philippines; Mumbai, India; Santiago, Chile; and Shanghai, China. These cities differ in geographical and climatic conditions; demographic characteristics;

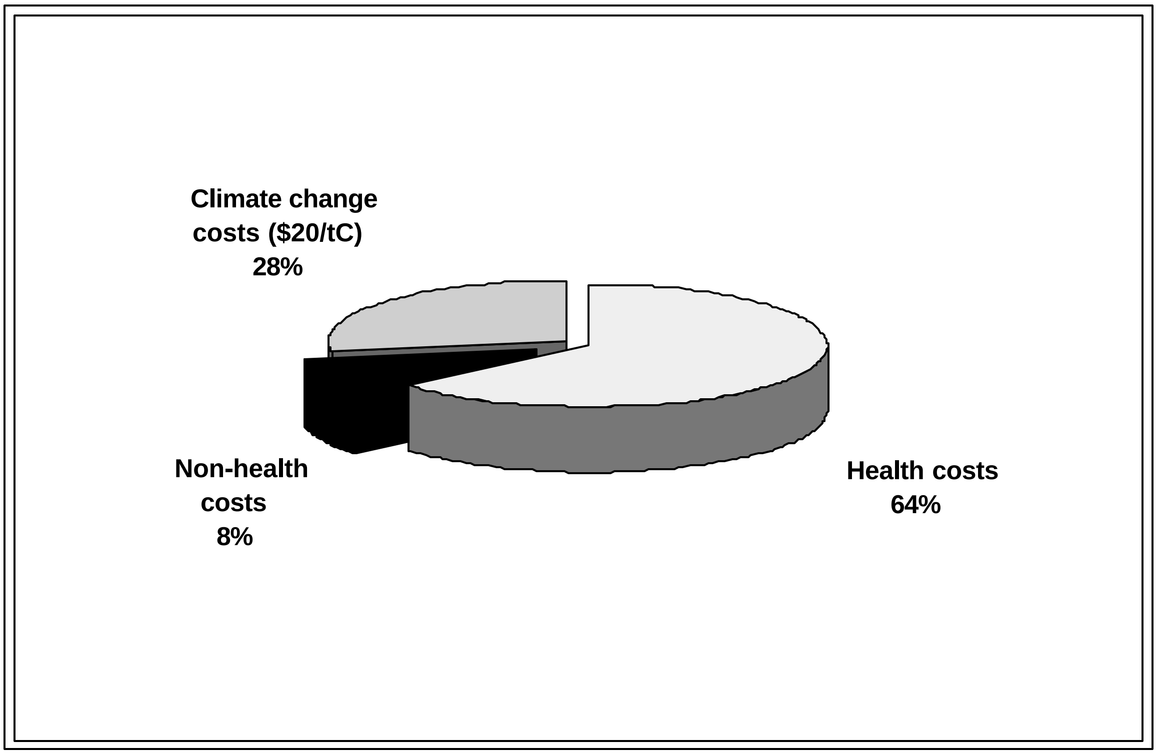
[2] Pre-1997 WHO guideline level

[3] Author's calculations. See text and Annex for explanation. The ambient levels of particulates for 126 cities are taken from the background materials for World Development Indicators, 1998; World Bank.

fuel mix and use patterns; sectoral composition; and income levels; and thus together represent a span of different factors affecting the magnitude of the environmental costs of various fuel uses. Therefore, the evidence emerging from this exercise is likely to be representative of the typical situation in many urban areas of developing countries.

The social costs of all environmental impacts assessed in the study reach US $ 3 billion, ***with health impacts being the largest portion of the costs for each city***. Chart 1 shows the shares of the health, 'local' non-health, and climate change impacts for the sample of six cities. Climate change impacts appear to be a major portion of non-health costs, but are less than half of the health costs imposed by fuel burning in urban areas.

Chart 1. The composition of environmental damages due to air emissions from fuel combustion in six cities

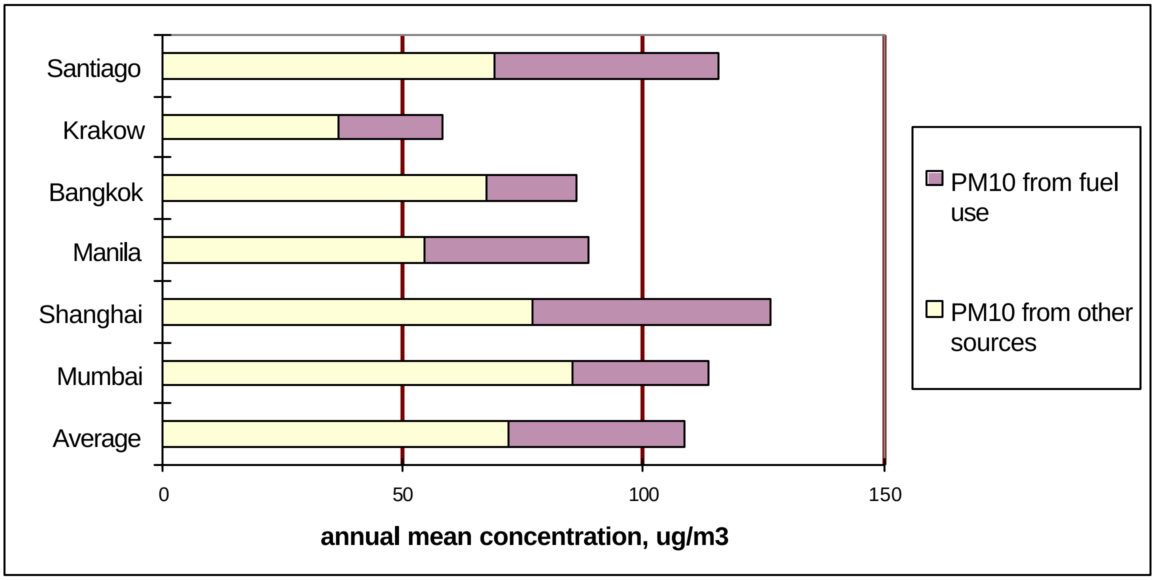


Source: World Bank estimates. See Lvovsky, et al., forthcoming.

Given that fuel combustion is not the only one, although significant, cause of the high level of particulates in cities (see Chart 2), overall health costs of poor air quality would be even greater.

The economic estimates of health damages are based on certain methodological tools, and are as credible as these tools are. The next two sections of the paper highlight a number of methodological issues raised by the valuing of health impacts of pollution. These issues naturally fall into two groups: (a) the actual identification and measurement of health impacts; and (b) estimating monetary values for associated morbidity (illness) and mortality (death).

Chart 2. Contribution of fuel use into the ambient levels of PM10



Source: World Bank estimates. See Lvovsky et al., forthcoming

3. Quantification of the health effects due to exposure to particulates: evidence from international experience and India

Dose-response studies for particulate matter. The most accurate way to measure the health impacts of air pollution in a given area is to conduct epidemiological studies for that area that 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) it may often be case that dose-response relationships established in other locations will have to be used instead. Moreover, the availability of other studies can be used to reduce individual study-level uncertainty. While one 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 reduce the uncertainty associated with individual study 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. The meta-analytic 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 which meta-analysis seeks to estimate. A number of meta-analytical reviews that have been presented in the literature (Ostro, 1994, 1996; Pope and Dockery, 1994; Schwartz, 1994) and served as the basis for extrapolating dose-response relationships to the situations where no specific epidemiological study has been done.

Over the past decade, over two dozen epidemiological studies have indicated an association between mortality and particulate matter. Some studies found correlations between mortality and other pollutants like sulfur dioxide or ozone. However, to the degree that different air pollutants tend to be correlated over time this might mean that different pollutants are used to explain what are essentially the same deaths several times over. To eliminate the possibility of double-counting, this paper only provides quantitative estimates of mortality effects related to particulate matter. This pollutant then serves as an potential index for many correlated pollutants. It is important to stress that particle effects on mortality have been observed in areas with both high and low sulfur dioxide and ozone concentrations, and in areas where particles peak at different seasons. Therefore, attributing the health effects to changes in particulates concentrations appears reasonable.

Application to developing countries. Most of dose-response studies have been conducted in industrial countries. This raises the question whether the extrapolation of the results for developing countries is valid. While some uncertainty remains, recent studies undertaken in less developed countries, such as Mexico City, Santiago, Chile and Bangkok, Thailand, lend support for this extrapolation. Also, the epidemiological studies typically provide information on the percent change in mortality due to a absolute change in ambient particulate matter. This may make the studies more appropriate for extrapolation since they predict changes relative to the baseline mortality rate, which may differ greatly between study areas. To determine the number of excess (prevented) cases of death due to exposure to higher (lower) concentrations of particulates, therefore, the percent change and the baseline rate in the impacted area must be determined.

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

I. Measures of particulate matter. The availability of data on PM10 and PM2.5 in both the original epidemiological study and the country in question is important for more credible results when applying dose-response functions derived elsewhere.

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 PM10) and to *fine particles* below 2.5 microns (PM2.5). The share of PM10 larger than 2.5 microns is termed coarse particles. TSP was the most common measure of particulates up to the 1980s, though some countries used what is called 'black smoke" which approximates quite well to PM10. The US-EPA shifted to a PM10 standard in 1986 after it had become apparent that high levels of TSP were often the result of wind-blown dust rather than pollution and had little impact on human health. Subsequently, it has been realized that even PM10 may contain substantial fractions of wind-blown dust -- illustrated by the fact that many of the places in the US which exceed the PM10 standard are dry, thinly populated areas.

Evidence from studies completed in the last 5-8 years suggests that it is *fine particulates* which are most likely responsible for the excess mortality and morbidity associated with high levels of exposure to particulates. Since most studies have used PM10 and not PM2.5 as their exposure metric (simply because PM2.5 has not been routinely monitored to date), this conclusion is based on several indirect but compelling factors. First, in most of the epidemiological studies finding associations between PM10 and adverse health, there is a high correlation between PM10 and PM2.5 and a low correlation between PM10 and coarse particles. Second, PM2.5 tends to penetrate indoors at a much higher rate than do coarse particles (or sulfur dioxide and ozone, as well). Third, fine particles will penetrate more deeply into the lung and are likely to be more reactive in the lung. A recent study (Schwartz, 1996) that included measurements of both PM2.5 and coarse particles in six U.S. cities found that daily exposure to fine particulates was strongly associated with premature mortality, while exposure to coarse particulates had little independent effect.[4] Though the weight of evidence indicates a greater concern for fine particles, a potential effect of coarse particles (the share of PM10 greater than 2.5 microns) cannot be completely ruled out.

Almost all fine particulates are produced directly or indirectly as a result of burning fuels. Industrial and other processes which produce large amounts of dust -- cement manufacture, mining, stone crushing, flour milling -- all tend to generate particles which are larger than 1 micron and most of this is larger than 2.5 microns. The species composition of fine particulates varies considerably across locations.

PM10 is a better proxy than TSP for fine particulates, and is employed in a variety of latest studies. A number of 'meta-analytical' estimates of mortality change have been produced across dose-response studies which use different measures of particulate matter - TSP, BS (Black Smoke), COH (Coefficient of Haze), to which standard conversion factors were applied. These estimates are less reliable, however, because variations in the levels of TSP or other measures of particulates may be quite different from those for PM10 and even more different for PM2.5, 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 PM10 can significantly reduce the uncertainty involved in converting from one measure of particulate matter to another. Therefore, studies using direct measurements of PM10 (or PM2.5) have been given primary attention in this analysis. ***To the extent that additional epidemiological studies can be undertaken, efforts should be made to site and utilize monitors that measure either PM10 or PM2.5.***

II. The existing pollution concentration. For most of the available studies, the mean concentration of PM10 is around 50 to 60 $\mu g/m^3$ with maximum values of around 150 to 200 $\mu g/m^3$ and consist largely of particles generated from combustion processes. Caution should be exercised in extrapolating these air pollution - mortality results to areas where the concentration or mix of pollution may be quite different. For example, in those cities in the developing world where (a) annual mean concentrations of TSP exceed 300 ug/m3, and (b) these high levels can not be easily explained by the pattern of fuel use, it is likely that a substantial portion of these particles are greater than 10 microns (PM10) and that the highest concentrations are driven by coarse, geologic particles. Calcutta and Delhi, India, illustrate this situation. Therefore, a direct extrapolation from the available studies over the whole range of concentrations may be misleading. This is well shown in the Cropper et al. study (1997) of air pollution mortality in Delhi. The study found that a change in mortality risk per unit change in TSP concentrations in Delhi is

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

significantly lower than in the US (see Table 1 below). In these cases, one option would be to apply the US-based dose-response functions only to a concentration up to a certain limit; say, of 200 $\mu g/m^3$ TSP for Delhi. Another option is to posit some non-linear function that begins to level off at some upper cut-off point.

Overall***, the need for more studies in developing countries, using advanced measurement techniques and taking a careful account for the specifics of the pollution situation, is obvious. India, where the ambient levels of particulates in many urban areas are extremely high and the mix of pollution is likely to be very different from industrial countries, should be a primary area for such studies.***

III. Disease-specific mortality profile. In some cases the distribution of deaths by cause may differ significantly between the country of interest and the countries where the original study was conducted. In this case, 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 make half of all deaths in the US. In Delhi, India, fewer than 20 percent of all deaths are attributable to these causes. Therefore, even identical reaction by susceptible groups of population in Delhi and the US to the change in the levels of particulates could result in a lower effect on total mortality in Delhi (Cropper and Simon, 1996). Thus, using dose-response estimates for respiratory and cardiovascular mortality and the associated local disease-specific mortality rates may provide better estimates of the air pollution impact on mortality since it better incorporates the local population at risk. Table 1 summarizes results for those studies, both in the US and abroad, that have considered disease-specific mortality. As indicated in the table, the percent change for total mortality as well as both cardiovascular mortality and respiratory morality are fairly consistent among the cities, even when results from developing countries are considered, except for the TSP-based Delhi study.

However, there are also some advantages of generating estimates using total mortality. One advantage is that it ensures that, based on the original studies, all mortality cases affected by air pollution are included in the dose-response function. The use of disease-specific approach in developing countries is often complicated due to limited access to disease-specific mortality data and the deficiencies in the death reporting system that may provide distorted information in respect to the actual causes of mortality. When this is a case, the use of all-cause mortality estimates may be preferred. If only cardiovascular- and respiratory-specific mortality are used in the dose-response function, the mortality effect may be underestimated if death certificates used in the original studies were misdeed or baseline rates in the country under study are incorrect. Finally, all-cause mortality approach is more suitable for a rapid assessment and cross-country comparison of the situation.

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 in local geography and climate, may not necessarily result in bias since these factors will be reflected in the crude mortality rate. For example, the percent increase in mortality per $\mu g/m^3$ of PM10 in Chile, with a much lower crude mortality rate, is similar to that found in many cities in the United States.

Table 1. Disease-specific mortality in alternative locations, percent change per 10 mg/m^3

City	First Author	Total	Mortality: Cardiovascular	Respiratory
Santa Clara, CA	Fairley (1990)*	0.8	0.8	3.5
Philadelphia	Schwartz (1992)*	1.2	1.7	3.3
Utah Valley	Pope (1992)	1.5	1.8	3.7
Birmingham	Schwartz (1991)	1.0	1.6	1.5
Steubenville	Schwartz (1992a)	1.1	1.5	--
Beijing	Xu (1994)*	0.7	1.45**	6.9***
Chicago	Ito (1996); Styer (1995)	0.6	0.4	1.4
Santiago	Ostro (1996)	1.0	0.8	1.3
Mexico City	Borja-Aburto (1997)*	1.0	0.92	1.65
Delhi	Cropper (1997)*	0.4	0.78	0.56
Bangkok	Ostro (1998)[5]	1.0	1.4	5.2

* Estimates are converted from TSP using ratio of 0.55

** Pulmonary heart disease *** Chronic obstructive pulmonary disease

Source: Adapted by B. Ostro in Lvovsky et. al., forthcoming

IV. 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 developed countries. While peak effects were observed among people of 65 and older in Philadelphia, US (Schwartz and Dockery, 1992b), in Delhi peak effects were reported in the 15-44 age groups. This implies more life years lost as a result of a death associated with air pollution (Cropper, et al., 1997). The latter study also shows that, though a change in mortality per 10 μg/m^3 change in TSP was found to be lower in Delhi than in the US, the number of life years lost for the exposed population of equal size appeared to be similar. This finding has important implications for valuation of the mortality costs that are discussed below.

Developing quantitative estimates for all-cause mortality from time-series studies. The epidemiological studies involve two principal study designs: time-series and cross-sectional. Time series studies correlate daily variations in air pollution with variation in counts of daily mortality in a given city and primarily measure the effects of acute exposure to air pollution. These studies are more common, and due to a progress in the several past years there are currently a sufficient number of studies for meta-analytical purposes that used PM10 as the actual measure of exposure.

Table 2 summarizes the evidence for nine PM10 studies, two of which were conducted in developing countries.[6] The pooled central estimate for the studies, relative to a 10 μg/m^3 change in PM10, is 0.84 percent.

[5] Presented at the Air and Waste Management Association Speciality Conference on Fine Particles, January, 1998, Long Beach, CA.

[6] In addition, a recent study of air pollution and daily mortality in Katowice, Poland, by Zejda et al. (1997), resulted in the central estimate of 0.7 percent change in total mortality per 10 ug/m3 change in PM10 levels,

Table 2. Estimated Percent Change in Mortality Associated with 10 $\mu g/m^3$ Change in PM10, Based on Studies Where PM10 was Measured

City	Author	Central	Low	High
Birmingham	Schwartz et al., 1991	1.0	0.2	1.5
Utah Valley	Pope et al., 1992	1.5	0.9	2.1
St. Louis	Dockery et al., 1993	1.5	0.1	2.9
Kingston, TN	Dockery et al., 1993	1.6	-1.3	4.6
Chicago	Ito et al., 1995	0.6	0.1	1.0
Los Angeles	Kinney et al., 1995	0.5	0.1	1.1
Santiago	Ostro et al., 1996	1.0	0.6	1.4
6-Cities	Schwartz et al., 1996	0.8	0.5	1.1
Bangkok	Ostro et al., 1998[7]	1.0	0.4	1.6
Weighted Average		0.84		

Source: Adapted by B. Ostro in Lvovsky et. al., forthcoming

Long-term exposure studies. However, these time-series studies only indicate the potential effect of short-term variations in exposure. This effect is likely to be smaller than that of long-term exposure. On the other hand, it is often asserted that the time-series studies to a large extent measure a so-called "harvesting effect", i.e. the deaths that are merely hastened by a few days, weeks or months due to high ambient concentrations of particular matter. A balance between chronic versus acute deaths and the actual extent of any "harvesting effect" can be assessed when the impact of long-term exposure is brought into comparison. Estimates of the impact of longer-term exposure are provided by cross-sectional and prospective cohort studies.

A common concern about cross-sectional studies is, however, whether all potential explanatory variables, such as variations between cities in smoking rates, diet, income, local industry, age distribution, etc., are adequately controlled. More confidence is placed on a second type of long-term exposure studies that involves a prospective cohort design in which a sample of population is selected and followed over time in each location. These studies use individual level data so that other health risks 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 exposures, smoking (present and past), alcohol use, age, and gender (Dockery et al. 1993 and Pope et al., 1995). Both of these studies report a robust and statistically significant association between exposure to particulate matter (measured as PM10, sulfate or PM2.5) and mortality.

To illustrate the effects of chronic exposure, the Pope et al. study is used since it has a larger sample size and more conservative estimates than the Dockery et al. study. When the empirical

which is consistent with Table 3.3. The study is not included in the meta-analysis because a complete report is not available in English.

[7] Presented at the Air and Waste Management Association Speciality Conference on Fine Particles, January, 1998, Long Beach, CA.

results for PM2.5 were converted to PM10 using a ratio of 0.65, a 10 $\mu g/m^3$ change in PM10 was associated with a 4.2 percent change in all-cause mortality.[8]

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. It also creates new challenges for valuing the health effects. The estimate that is based on the chronic studies or combines the results of the acute and chronic studies implies a long-term exposure to air pollution, and thus can not be used for assessing the short- or medium term impacts of an annual change in urban air quality without a significant adjustment.

The calculations of the mortality effects due to exposure to ambient PM10, given in this paper, use the value of 0.84 percent per 10 ug/m3. Given the balance of evidence across various studies, it should be regarded as a rather conservative estimate for the impact of such exposure with respect to the ambient levels of PM10 attributed to fuel burning or similar pollution mix. Further, Tables 1 and 2 show that the results from studies for Mexico City and Beijing, even when based on TSP measurements, as well as PM10-based results for Bangkok and Santiago, are very consistent with evidence from industrial countries.

However, from this discussion and the results of the Cropper et. al 1997 study of the air pollution and mortality in Delhi***, India appears to be a place which requires a greatest care in applying dose-response relationships derived from studies elsewhere.*** The question arises whether and under which circumstances can we justify the use of a higher, 'meta-analytical' estimate for Indian cities? This issue will be addressed for the case of Mumbai in section 5 of this paper. Meanwhile, it should be noted that in generating international comparison estimates for 126 cities, this dose-response relationship was not applied to the whole range of monitored concentrations (commonly measured as TSP and converted to PM10, using 0.55 ratio), but only to concentrations up to 100 ug/m3 PM10 for India, as recommended above. For Delhi, this approach would result in even fewer deaths as compared to using a much lower mortality risk from the Cropper et al., 1997 study over the reported multi-year TSP average level of 375 ug/m3 (concentrations above the respective pre-1997 WHO guidelines were used in both cases).

Morbidity. Dose-response functions are also derived for many lesser health impacts, such as respiratory hospital admissions (RHA), cardiovascular hospital admissions (CHA), emergency room visits (ERV); chronic bronchitis (CB); bed disability days (BDDs), restricted activity days (RAD), asthma attack (AAs), acute respiratory symptoms, and lower respiratory illness in children (LRI). There are far fewer dose-response studies for morbidity end-points due to exposure to air pollution that makes the available meta-analytical estimates less robust as compared to the mortality effects (for some health end-points, the morbidity estimates 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 due to air pollution. The largest portion of the morbidity costs falls on new cases of chronic bronchitis, which, according to some studies, even exceed the economic costs of premature death due to air pollution (see, for example, US EPA, 1997). Therefore, ***more epidemiological studies quantifying these effects are needed, especially in countries like India and China whose***

[8] For comparison, the Dockery et al. study yields 9.5 percent change in all-cause mortality per 10 ug/m3 change in PM10. One of possible explanations may be that the two studies succeed in accounting for cumulative exposure to particulate matter to a different extent due to the greater duration of the Dockery et al. study.

urban residents suffer from the highest levels of exposure to particulates. The mortality and morbidity effects employed in this paper are summarized in Table A.1 provided in the Annex.

Lead and ozone. The epidemiological work is not yet sufficiently advanced to provide robust dose-response functions for all health effects of particulate matter or some other pollutant-health combinations. This paper focuses on particulate matter - the most serious problem in India cities - and does not consider the health effects of ozone or lead. For ozone, most of the available studies indicate an acute and reversible effect on respiratory illness. There is also the possibility that long-term exposure to ozone, especially for children exposed for long periods of time during the day, may result in chronic inflammation and premature aging of the lung. The ambient measurements of ozone are largely not available in India. While the indirect data (such as the measured levels of NO2) indicates that the ozone levels are unlikely to be high at the moment, the situation may change over time with the increased volumes of traffic and higher octane specifications of gasoline. Thus, the necessary monitoring capacities should be developed. Previous studies have indicated that the effects of ozone, relative to particles, is small (Krupnick and Portney, 1991). Lead effects, on the other hand, are likely to be large in terms of neurodevelopment effects on children and cardiovascular effects on adults. The removal of lead from gasoline is an important and unambiguous issue. Many countries around the world, including Asia, has either already accomplished or committed to this task, and India should clearly follow this path. Certain industries nearby residential areas is another significant source of exposure to lead in India and other developing countries.

In sum, ***the reduction in the exposure to particulates and lead, as well as epidemiological studies needed to build sufficient scientific and political support for these action, are of immediate priority for India.***

4. Economic valuation of the health impacts due to air pollution

Valuation of a statistical life: general approaches and challenges. Everyday individual actions in which people trade money against a small reduction in personal safety can be used to infer the *value of a statistical life (VOSL)*. This is not the same as valuing an actual life, and should not be interpreted as such. Instead it involves valuing ex-ante changes in the level of risk people face and then aggregating them. Since the exact identity of those at risk is unknown, valuing ex-ante changes in the level of risk is the appropriate policy context.

The literature on the VOSL, or *Willingness-To-Pay (WTP) to avoid a statistical premature death*, is relatively well-developed and there exist several analyses in which the empirical estimates, mainly from the US, are reviewed, such as Fisher et al. (1989), Miller (1990), Viscusi (1992, 1993) and TER (1995). The two most complete surveys of the existing literature suggested a mean VOSL of US$ 3.6 million (IEI, 1992) to US$ 4.8 million (US EPA, 1997) in 1990 dollars.

There is also a substantial literature on the valuation of life that relies on so-called *'Human Capital' approach*. Human capital is the present value of future labor income. The human capital and the WTP approach are not entirely unconnected. More specifically, 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 human capital mortality cost estimates in Table A.3 in Annex with the WTP estimates of US$ 3.6 - 4.8 million). Seemingly straightforward, the application of human capital approach to developing countries can still be problematical due to distorted wages, cross-subsidization of public services, difficulties with valuing various homemaking services, high unemployment rates, etc. 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.

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

There are several uncertainties which complicate the transfer of available WTP estimates into the context of premature deaths caused by air pollution in developing countries. A set of problems stem from the fact that the existing results refer almost exclusively to lives lost as a result of accidents at work rather than air pollution. More specifically, it is argued that remaining life-years of those who die in occupational accidents is much greater than those who die as a result of poor air quality. Further it is argued that those who are most at risk are already suffering from some underlying condition that may affect the values to ba attached their lives. It is also argued that the contextual effects are important and finally there is the issue of latency to consider. Finally (and most importantly in quantitative terms), income levels differ greatly between the surveyed populations and the 'target' populations of the developing countries that requires a significant adjustment in the US-based VOSL. Since the assumed VOSL determines the damage cost estimates which emerge from air pollution studies, these issues should be carefully interpreted in the approaches adopted to placing a monetary value on the health outcomes of exposure to air pollution.

Age effects, underlying health conditions and social costs. If age effects are important in determining 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, then the effect of applying these VOSL estimates to the air pollution context will introduce a bias. Labor market studies, upon which the VOSL estimates are usually drawn, measure compensation for risk of instaneous death for people of about 40 years old and thus value approximately 35 years of life (Viscusi, 1993). The study of Philadelphia in the US found that the excess mortality due to air pollution almost entirely falls on the age group of 65 and older (Schwartz and Dockery, 1992b), and other studies that utilize age-specific mortality (except for Cropper et al., 1997) indicate that the vast majority of deaths related to higher concentrations of particulates occur in the over-65 age category (Fairley, 1990; Saldiva, 1992; Ostro et al., 1996; 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 age distribution of those involved in WTP studies and those primarily affected by pollution should change the respective estimate of VOSL.

A possible approach was 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. Comparison, for example, of the remaining years of life for the average respondent of labor market studies and the average person from the age group of over 65 in the USA (35 and 10 years lost, respectively) at a 10% discount rate gives an adjustment factor of 0.64.

A particular benefit of this 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, highlighted by the Cropper et al. (1997) study of air pollution in Delhi. The study found that, though mortality risk due to exposure to particulates (measured as TSP) in Delhi considerably lower than in the US, the respective number of life years lost is similar to that in the US. This result is not merely coincidental - a greater number of life-years lost per an average death from air pollution occurs precisely due to 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 PM10-based mortality studies, as suggested in the previous section, in combination with the VOSL adjusted for a number of life years lost will result in a more robust assessment of the mortality costs in cases like Delhi.

What is further important and advocated in this paper, is the need for aligning the economic approaches to valuing sickness and premature death with the concept of Disability-Adjusted Life Years (DALYs). DALYs are a standard measure of the burden of disease (WDR 1993; Murray and Lopez, 1996) that 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 that are usually given to the illness and premature mortality at different ages. Thus, it is possible to link the VOSL obtained from labor market studies with the corresponding number of DALYs lost in order to estimate the implicit value per DALY, and then to adjust the respective VOSL according to 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 US labor market studies that represent people of around 40 years old corresponds to 22 DALYs lost while an average death of 65 year old (assumed to be a mean age of those fatally affected by particulates) corresponds approximately to 10 DALYs lost. This implies that a value per DALY in the US is $ 164,000 and the WTP 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 as opposed to 64 percent based on a simple discounting of life years lost at 10% rate. The reasons for such difference are: (a) in using a much lower discount rate while calculating DALYs; (b) in the different social values assigned to a year of life at different ages; and (c) in the different weights given to the healthy years and years lived with disability, whose portion in the total years lost due to premature death increases at older ages. The incorporation of the latter factor in the DALY measure is important because it addresses another issue in the debate over the relationship between the mean VOSL and the value of an average death caused by air pollution; namely, the WTP 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 a number of acute deaths from exposure to particulates merely represent 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+ 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 for a definite answer; however, the difference is unlikely to be near as substantial as for the mean VOSL.

Contextual effects, latency effects, and the valuation of changes in life expectancy. It is generally accepted that the value that individuals place on the avoidance of risk depends upon 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 the job or transport related risks, so these differences should certainly be considered when trying to transfer existing value of life 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; thus, reducing the exposure of the individual up to the point where 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 uncontrollable risks. The extent to which this under-values the cost of air pollution is uncertain.

Another important characteristic of air pollution is that it often presents latent rather than contemporaneous risks. Cropper and Sussman (1990) convincingly demonstrate that the willingness to pay for a reduction in future risks are to be discounted at the consumption rate of interest. An additional complexity, however, is that it may be difficult to separate out issues relating to the quantity of life from those relating to the quality of life for latent risks. 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 WTP to reduce these sorts of risks would be rather greater than to reduce risks of a death following an automobile accident.

This issue of latency has particular importance to air pollution studies when one considers the findings reviewed in the previous section that the majority of premature deaths from particulate concentrations were 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 of valuing "harvested" deaths from the acute exposure studies has led to a search for another approach to measure the impact of air pollution on human health and mortality risk. Such an approach can be seen in quantifying and valuing changes in life expectancy of the exposed population caused by variations in the air quality. This approach deals with both chronic effects and the "harvesting" effect by making comparisons between the average life expectancy of individuals exposed to different concentrations of particulates over a long term.

The life expectancy approach involves (Thurston et al., 1997): (a) estimating the change in life expectancy by age group implied by the change in ambient particulates; (b) establishing a WTP for the change in life expectancy by age group; and (c) multiplying these two values with each other and by the population in each age group, and adding up. The major problem here is the lack of empirical evidence regarding a WTP for an increase in life expectancy. Currently, only one study (Johannesson and Johansson, 1996) conducted a contingent valuation survey in respect to changes in life expectancy, with a large number of uncertainties attached to it, so that an extensive further research is needed.

Generally, ***the approach to valuing changes in life expectancy as a result of long-term exposure to air pollution seems very promising***, not only because it addresses the uncertainties of adjusting the WTP to avoid contemporaneous risks at the prime age to the air pollution context, but also due to a high political sensitivity of the VOSL concept. It may be more politically acceptable to explicitly incorporate the value of a change in average life expectancy in the design of environmental policies than the VOSL.

Valuation of acute morbidity effects. Air pollution also affects human morbidity, and the valuation of illness and disability is very important to assessing the social costs of air pollution and cost-benefit analysis of control measures. The literature on *WTP to avoid the morbidity effects* is very limited in scope and based entirely within the United States. An alternative, 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, which is 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, 1982). Although the COI approach is often viewed as easily applicable to any country, subsidized and/or inadequate medical services and drug supply in many developing countries make it difficult to calculate the economic costs of health care. More importantly, COI will underestimate WTP because it fails to account for the disutility of illness. Since the disutility of illness is likely to be a major component of WTP, the COI approach cannot ever be entirely satisfactory. As a result, most preceding 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 it is not.

One approach that has emerged to deal with the paucity of WTP literature and the inadequacy of the 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 cardinal scale from 0 (death) to 1 (perfect health). Any health state can be evaluated by considering its impact upon various symptoms, its effect upon social activity, physical activity and mobility, and its duration. By these means, the conceptually appropriate WTP values can be obtained for each and every 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. In making such extrapolations, it is important to distinguish between acute effects and chronic effects, because the very fact of irreversibility of a poor health state adds a significant component to WTP for avoiding this health state, that will not be captured by WTP estimates to avoid temporary acute disorders. This approach has been taken in a paper by TER (1996) and is followed in this paper. Table A.2 in the Annex contains the adopted base valuation parameters. It should be noted that the WTP estimates are consistent with and rather close to the COI estimates, available for some morbidity outcomes.

Valuation of chronic bronchitis. Chronic bronchitis (CB) is the most severe morbidity endpoint, for which the dose-relationship is established (Abbey et al, 1993), that may last from the beginning of the illness through the rest of the individual's life. Therefore, the valuation of this illness should be done separately from the other morbidity effects, related to air pollution.

There are two studies that provide estimates of WTP to avoid chronic bronchitis, using the contingent valuation analysis (Viscusi et al., 1991; and Krupnick and Cropper, 1992). Based on these studies, the recent US EPA review of the costs and benefits of cleaner air (USEPA, 1997) recommends the mean WTP of US $ 260,000 (in 1990 dollars). This is regarded as a reasonable value relative to 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 (from 30 to 60 year old). It is, however, important to keep consistency in a ratio between the VOSL and WTP to avoid a chronic illness. Since the US EPA 1997 report uses the VOSL of US $ 4.8 million while this study adopts a lower estimate of US $ 3.6 million, we downsized the WTP to avoid a new case of CB accordingly and used the base value (before an adjustment for income) of US $ 195,000 in our calculations.

Income effects. One of the fundamental issues of valuing the reductions in risk is that the WTP rises with income. Given that the existing VOSL estimates are taken almost exclusively from the US there is a clear need to adjust the VOSL for income effects before applying the results to developing countries. The literature on the income elasticity of WTP for reducing the risk of insults to health however is extremely limited. A simple average of the three available studies yields an income elasticity of 0.7 (Jones-Lee et al., 1985; Biddle and Zarkin, 1988; Viscusi and Evans, 1990).

It is important to note, however, the acute sensitivity of the social costs of ill-health to the value of this parameter. The difference in the income adjustment for India between the use of elasticity of 0.4 or 1.1 is nearly 20 times. To maintain a degree of conservatism in this valuation exercise, a higher income elasticity of 1 for both the VOSL and morbidity cost estimates is used for all calculations in this paper. The finding that the income elasticity of demand for medical goods and services is shown, by cross sectional analysis of per capita expenditures in the 1980 International Comparisons Project, to be 1.05 lends support to this decision[9].

Key messages and observations. The analysis of methodological issues highlights that future work intended to reduce the uncertainty associated with the estimates of the economic costs due to air pollution should focus on ***determining the values of morbidity and mortality impacts in India and other developing countries***. Alternatively, in many cases the aggregate measures like DALYs that do not involve the direct costing of the health effects due to air pollution can be used for ranking the priority areas and mitigation options.

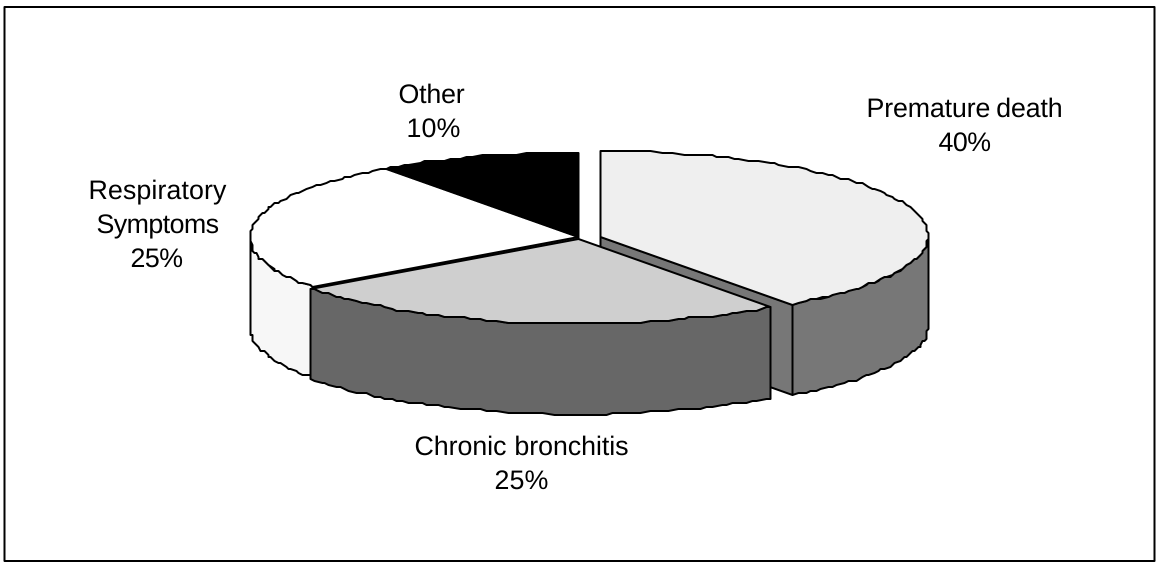
An important observation from this review is ***an increasing convergence between the approaches to assessing the burden of ill-health being devised by economists and public health specialists***. This is evident from both: (a) an attempt to combine the measure of DALYs with the age- and context-specific VOSL; and (b) integration of the WTP to avoid illness with the health-status index. This tendency should be strongly supported as it serves to promote a greater acceptance of the aggregate measures of the burden of disease, provide for consistent assessment

[9] There is also an issue on whether an income in developing countries should be measured in US dollars at a market exchange rate or using PPP (purchasing power parity) conversion rate while transferring the VOSL estimates from the rich countries. In this study, a market exchange rate is used. Without further argument about the merits or shortcomings of each approach, it is worth noting that the use of PPP-based estimates, advocated in a number of studies (e.g. Thurston et al., 1997; Markandya, 1997), would considerably increase the social costs of mortality and morbidity, especially in the lower income countries like India or China.

of environmental health priorities and unite public efforts to reduce the risk of exposure to environmental hazards.

When different health end-points of air pollution exposure are brought to one denominator through the valuation exercise, premature deaths account for about 40 percent of the health costs and various illnesses provide for the larger 60 percent. Chronic bronchitis and acute respiratory symptoms are the largest contributors to the economic costs, associated with morbidity. Chart 3 details the composition of the air-pollution related health costs by cause as based on the assumptions of this analysis. ***This, again, points to the need for more studies that would assess and value major morbidity outcomes***.

Chart 3. A typical composition of the health costs due to air pollution by cause



Source: World Bank estimates. See Lvovsky et al., forthcoming

A dominant share of the social costs of sickness in the total health damages due to air pollution can be used to strengthen the dialog with policy makers, as it reduces the reliance on arguments that are surrounded by the controversy of valuing a statistical life. Also, the portion of these costs that represents morbidity closer relates to economic losses in productivity. If the costs of acute mortality together with one-fifth of the CB cost are taken as a rough proxy for such losses, then the productivity impacts would be somewhat 40 percent of the total health costs. For 12 largest Indian cities, it implies annual productivity losses at the magnitude of US $ 800 million (with the total social costs of ill-health of US $ 2 billion per year).

5. Health impacts and priorities for pollution control: a case of Mumbai

Mumbai is one of the urban agglomerations in the six cities study, mentioned above, and this section will discuss the results of the analysis for this city with a particular objective to illustrate how the assessment of the health impacts can be used for setting pollution control priorities.

As part of the study, a special model has been developed that links these impacts to: (a) emissions from various economic sectors or sources, and (b) fossil fuel use in each sector. The five most damaging fuels -- coal, fuel oil, diesel, gasoline, and wood -- were examined. The study was designed as a rapid cross-country exercise and was intended to analyze the evidence for the six cities as a whole rather than the details for each individual city. This should be kept in mind during the discussion of the results for Mumbai that follows. The magnitude of damages in the absolute terms will be true only to the extent to which the assumed health effects and economic values match the Mumbai conditions; however, the relative priorities across pollution sources do not depend upon these assumptions (they depend, though, on the validity of the standardized dispersion model and emission inventory, but not to the extent that may reverse the broad conclusions).

The total annual health damages from combustion of various fuels in Mumbai, based on 1992 inventory, amounted to US $ 150 million. Table 3 shows the shares of three major groups of combustion sources in these damages: vehicles; large power utilities and industries; and small boilers and stoves used by small-scale industries, commerces and households, as well as details the sectoral composition of damages for specific health effects.

Table 3. Mumbai: The health impacts of fuel use by a category of combustion sources

City	All sources	*Power plants& large boilers*	*Small boilers&stoves*	*Vehicles*
Cases:				
Premature death	2,140	175	1,442	523
Chronic bronchities	7,796	637	5,255	1,905
Respiratory symptoms day	34,036,340	2,780,621	22,940,122	8,315,598
Restricted activity day	10,694,478	873,692	7,207,962	2,612,824
Social costs, '000 US $:				
Premature death	71,601	5,849	48,258	17,493
Chronic bronchities	31,396	2,565	21,161	7,671
Respiratory symptoms	30,928	2,527	20,845	7,556
Restricted activity	11,706	956	7,889	2,860
Other effects	1,609	131	1,084	393
Total	**147,240**	**12,029**	**99,238**	**35,973**
per resident, US$/psn.	12	1	8	3
as a share of income, %	3%	0%	2%	1%
a share by source, %	**100%**	**8%**	**68%**	**24%**

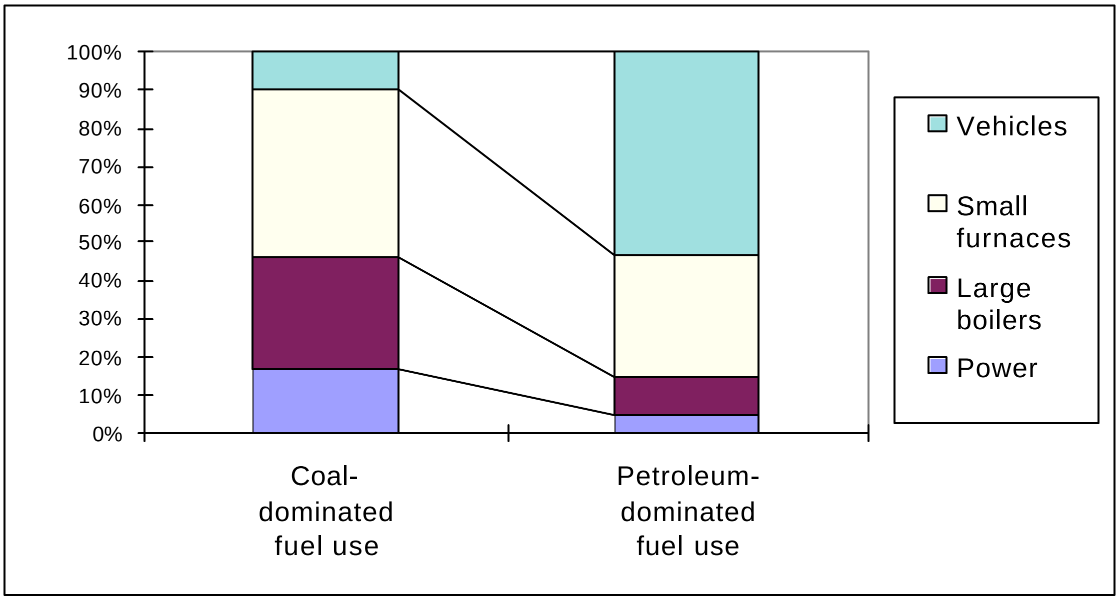
Source: Author's calculations.

Turning back to ***a question of what type of a dose-response relationship for mortality risk should be used for Mumbai*** - a meta-analytical estimate from a series of PM10 studies elsewhere or a value from the Delhi study, it should be noted that the health impacts in Table 3 reflect only an increase in the levels of PM10 (30 ug/m3 annual agglomeration-wide average) that is attributed to the emissions from combustion of various fuels. These are not the impacts of the overall exposure to the ambient levels of particulates which would be greater. ***Because both the pollution mix from fuel burning and a corresponding range of PM10 concentrations match very well the situations in industrial countries where most dose-response studies were undertaken, the use of a meta-analytical estimate for the mortality effect is better***

justifiable, and in combination with the DALY-adjusted VOSL should not bias the respective social costs.

The assessment shows that most of the health damages come from vehicles and small non-mobile sources. The largest contribution from small boilers and stoves is due to a relatively wide use of highly polluting fuels, such as wood, coal and heavy fuel oil. ***The cross-country analysis reveals two principal patterns of sectoral composition of health and overall environmental damages from fuel use***, illustrated by Chart 4. Where coal and wood are widely used by small sources, these sources of air pollution typically account for the bulk of the damages (further exacerbated by indoor air pollution from these fuels). Once households and small businesses switch to cleaner and more convenient fuels, like LPG, kerosene, etc., which is usually coincides with an increase in traffic volumes, transport becomes the major problem. ***Mumbai and many other India cities are currently in transition from one pattern to another.***

Chart 4. Two typical patterns of sectoral contribution to the environmental costs of fuel use



Source: World Bank estimates. See Lvovsky et al., forthcoming

Priorities for air pollution control should focus on measures that provide largest benefits at a given cost. Reduction in exposure and associated improvements in health constitute the major portion of the environmental benefits. Table 4 indicates the relative magnitude of the health benefits that can be achieved by various control options applied to different pollution sources. The most significant benefits are to be brought by measures that promote the conversion of small-scale industries and households from coal and wood to cleaner fuels. Implementing vehicular inspection and maintenance programs would also lead to substantial improvements in pollution levels and health benefits.

Table 4. Mumbai: Health benefits from pollution control options at different sources

Options:	Reduction in: Total health costs	Premature death	Respiratory symptoms	Chronic bronchitis
	000' US $	*cases*	*days*	*cases*
Switching small sources from coal to light oil/LPG	56,256	821	13,059,652	2,991
Switching small sources from wood to light oil/LPG	13,288	196	3,126,987	716
Inspection & maintenance program for vehicles	11,906	173	2,752,227	630
Reducing sulfur content of fuel oil to 0.5 %	10,395	154	2,458,210	563
Reducing sulfur content of diesel to 0.25 %	8,175	119	1,889,646	433

Source: Author's calculations.

The next step in priority setting involves the comparison of the benefits with the costs of control options. This exercise has been done for Bangkok, another city from the six cities study, and is used here merely for illustration of how the assessment of health impacts can be integrated in the process of developing a cost-effective pollution control program. Bangkok is a typical example of the urban air pollution pattern dominated by vehicular emissions and the use of petroleum. In this city, the least cost program for reducing the exposure levels of PM10 that are specifically linked to fuel use include the following options (listed in order of their cost-effectiveness in terms of mitigating the adverse health effects):

⇒ Fuel switching (heavy fuel oil to gas) for industrial/commercial boilers and power plants
⇒ Use of gasoline rather than diesel for light duty pickups, trucks, etc.
⇒ Replacement of 2-stroke motorcycles by 4-strokes with stricter emission standards
⇒ Installing new diesel engines in buses and trucks to meet stricter emission standards (equivalent to those proposed by the EU for 2000 onwards).

6. Summary of issues: the need for integrating health and environmental policies

The ten main policy conclusions can be drawn from the analysis presented in the paper:

- The urban population of India bear a high burden of the health costs due to exposure to particulate matter by international comparison that requires immediate actions and targeted environmental policies

- The health costs imposed by air emissions in urban areas of India and many other developing countries exceed other environmental damages, including the potential impact on global climate change, by large margin; therefore, in the short and medium term, the primary attention should focus on mitigating these costs, utilizing synergies between local and global issues when possible

- The development of the monitoring system that will provide reliable and adequate information about the levels and major sources of exposure to most damaging pollutants (such as PM10, PM2.5 and, eventually, ozone) is necessary for understanding the problems and designing effective interventions; the pay-off will be high due to substantial cost savings through focusing on priority areas and sources

- India appears to be a country where a greatest care is required while applying dose-response relationships for particulate matter derived from studies elsewhere; thoroughly designed local epidemiological studies using monitors that measure PM10 or PM2.5 and tracing the impacts of long-term exposure should be supported

- Valuation of illness and premature death in the context of India and other developing countries is an issue that needs more attention from the international community of economists and supporting institutions

- Given that the social costs of illness due to air pollution may be even more significant than the costs of premature death *per se* (for which the preceding illness is often either a determining or compounding factor), more work is needed that would assess and value the morbidity outcomes

- An emerging tendency for a greater convergence between the economic and public health approaches to assessing the burden of diseases from air pollution causes is a positive sign that should be further encouraged

- In a country like India where resources are very scarce relative to a variety of development objectives, it is of special importance to carefully set air pollution control policies and priorities according to the health impacts and anticipated benefits

- Advances in the areas of environmental health and economics make this approach possible; the major challenges are to mobilize political support and overcome institutional constraints

- Combining efforts of various experts and institutions that work in the overlapping areas of public health, environment and economics can significantly contribute to influencing the policy makers and the public, and leverage decisions that bring large environmental health benefits

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Annex

Table A.1: Dose-response functions for PM10 and SO2 used in the paper (per 1 ug/m3 change in the annual mean level)

Health effects	***PM10***	***SO$_2$***
Mortality, percent change in all-cause mortality rate	0.084 (Ostro, 1998)	
Chronic Bronchitis / 100,000 Adults	6.12 (**3.06**; 9.18)* (Ostro, 1994)	
RHAs / 100,000 People	1.2 (Ostro, 1994)	
AAs / 100,000 Asthmatics	6,499 (Maddison, 1997)	
ERVs / 100,000 People	23.54 (Ostro, 1994)	
RADs / 100,000 Adults	5,750 (Ostro, 1994)	
LRIs / 100,000 Children	169 (Ostro, 1994)	
Respiratory Symptoms / 100,000 Adults	18,300 (Ostro, 1994)	
Cough Days / 100,000 Children		1.81 (Ostro, 1994)
Chest Disc. Days / 100,000 Adults		1000 (Ostro, 1994)

* Value of 3.06 (low estimate) is chosen for this exercise.

Table A.2. Base values of health effects used in the paper

Health Status	***Monetary value per case (WTP-based), 1990 USD***
Premature death due to air pollution	1,620,000
Chronic Bronchitis	195,000
RHAs	4,225
AAs	63
ERVs	126
RADs	53
LRIs	44
Respiratory Symptoms	44
Cough	44
Chest Discomfort	50

Source: US EPA, 1997; TER, 1996; and Lvovsky, et. al, forthcoming. Values are for the U.S. income level in 1990 and are adjusted for individual cities according to the income difference with the U.S. 1990 GDP per capita.

Table A.3: Human capital and mortality cost by age in the USA

Age Group	Life Years Lost	Mortality Cost
Under 5 years	75	$502,421
5 - 14 years	68	$671,889
15 - 24 years	57	$873,096
25 - 44 years	42	$785,580
45 - 64 years	25	$278,350
65 years and over	10	$22,977
All Ages	12	$143,530

Source: The Institute for Health and Aging. Note that the cost estimates are based upon life expectancy at the time of death and include labor force participation rates, average earnings, the value of homemaking services and a 6% discount rate with which to convert figures into their present value counterparts. All figures are in 1992 prices