VALUATION OF HEALTH IMPACTS OF AIR POLLUTION FROM POWER PLANTS IN ASIA: A PRACTICAL GUIDE

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Valuation of Health Impacts of Air Pollution from Power Plants in Asia: A Practical Guide[1]

ABSTRACT

Assigning a monetary value for air quality reduction and associated health outcomes of electricity generation is both difficult and essential; it is difficult because methods are cumbersome, data intensive and costly, however dollar value of cost of air pollution is imperative for formulating pollution control policy. From a practical point of view, complete and detailed studies for every power plant project is not feasible. This paper reviews Impact Pathway Approach (IPA) for valuing health costs of air pollution and recommends a streamlined methodology combining site specific studies and benefit transfer for quick assessments. The paper also illustrates the proposed methodology by applying it to an 800 MW coal-fired power plant in India. Results show that pollution abatement is economically efficient; total health cost of air pollution can be reduced to $1.05 cents per kWh from $12.58 cents per kWh with pollution abatement cost of $0.28 cents per kWh. Strengthening available regulatory measures of pollution control and implementing a rigorous monitoring program to ensure installation and use of pollution control equipment is therefore welfare improving.

JEL Classification: Q40, Q51, Q53, I10

Keywords: Air Pollution, Impact Pathway Approach, Dispersion Modelling, Dose Response Function, Economic Valuation, Benefit Transfer

[1] We would like to thank the anonymous reviewer for comments on the earlier draft of the paper.

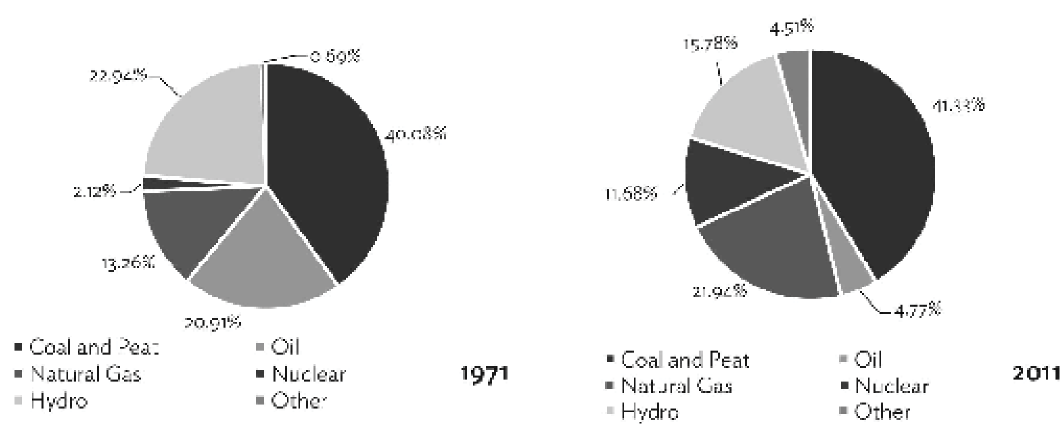
ABBREVIATIONS

APED	–	Air Pollution Epidemiology Database
BT/VT	–	Benefit Transfer/Value Transfer
COI	–	Cost of Illness
CBR	–	Chronic obstructive respiratory disease
COPD	–	Chronic-obstructive pulmonary disease
CRF	–	Compromised respiratory function
CVHA	–	Cardiovascular hospital admission
CV	–	Contingent Valuation
DRF	–	Dose Response Function
ERV	–	Emergency Room Visits
EXTERN-E	–	External Costs of Energy
HCA	–	Human Capital Approach
NO	–	Nitrous Oxide
HEI	–	Health Effects Institute
IPA	–	Impact Pathway Approach
PAPA-SAN	–	Public Health and Air Pollution in Asia–Science Access on the Net
PM	–	Particulate Matter
RAD	–	Restricted Activity Days
RHA	–	Respiratory Hospital Admission
SO	–	Sulfur Oxide
TAPM	–	The Air Pollution Model
VoSL/VSL	–	Value of Statistical Life
WLD	–	Work loss days
WTP	–	Willingness to Pay

1. INTRODUCTION

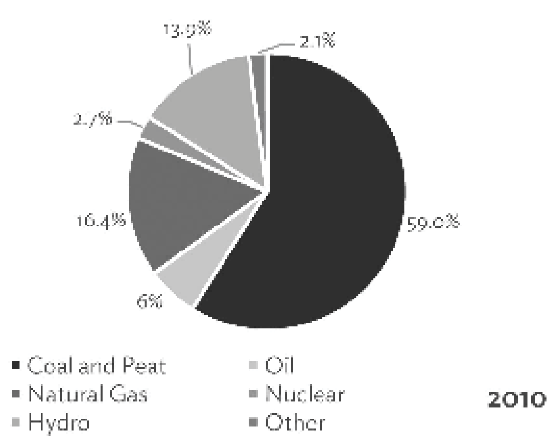
Over a 40-year span, since the early 1970s, fossil fuels remain as the dominant source of electricity; constituting about 75% of world electricity generation in 1971 and 68% in 2011 (Figure 1). Coal remains as the single most important energy source for electricity generation. The dominance of coal is more visible (59% in 2010) in South Asia's power sector (Figure 2). Renewable energy sources show limited penetration despite gaining footing in the market. Given that about 400 million lack access to power, while of those with access in the region continue to face power shortages, power generation in South Asia is projected to increase by about 130% between 2010 and 2020 (Rahman et.al 2011). This brings in one important aspect–environmental pollution associated with increased power generation– which is often overlooked. Burning fossil fuels create both local and global air pollutants (mainly Carbon Dioxide). This paper focuses on local air pollutants. Often, lack of information on social welfare loss due to air pollution hampers efforts to reduce air pollution. This paper undertakes a methodological review on estimation of health cost of local air pollutants from power generation and recommends a pragmatic approach for quick assessments.

Figure 1: World Electricity Generation by Energy Source



Source: OECD. 2014. OECD Factbook 2014: Economic, Environmental and Social Statistics.

Figure 2: South Asia Electricity Generation by Energy Source



Source: OECD. 2012. OECD Factbook 2012: Economic, Environmental and Social Statistics.

Conventional energy sources such as coal, diesel, and furnace oil come with significant costs to one's health and welfare. Inadvertent consequences such as these are generally seen as externality. When externalities are involved there is a divergence between private and social cost; social costs such as health cost of air pollution are borne by society at large. Widespread recognition of such costs is essential in order to nudge or steer pollution control policies in a desired direction. From a practical perspective, internalizing externality costs becomes essential when 'avoided health costs' are recognized under the benefit stream of a cost–benefit analysis (CBA) of renewable energy projects, which comprise the long term solution for local and global pollution problems of power generation. Of the renewable energy sources, solar provides greater physical potential compared to wind and hydro but capital costs of solar tend to be higher relative to conventional energy sources such as coal. Offsetting high capital costs of renewable energy sources such as solar may require incorporation of avoided costs of both local and global pollutants in CBA.

This paper uses the impact pathway approach (IPA) to quantify health impact of power generation. The IPA is built upon four main steps: (i) site specification and emissions estimation; (ii) quantification of ambient pollution concentrations through dispersion modeling; (iii) quantification of health impacts resulting from changes in ambient concentration; and (iv) valuation of health impacts in monetary terms. Each step either represents a challenging modeling and data issues or require knowledge on environmental valuation methods. The valuation methods come with strengths and weaknesses, sometimes resulting to a skepticism about accuracy of the results. Building on existing research in this area, this paper provides some relief to practitioners by first, presenting a practical guideline including a combination of original studies (where data can be readily available and methods are relatively simple) and benefit transfer (where original studies are not feasible with time and resource limitations and reasonable amount of research findings are already available).

Generally the original site specific studies are preferred over benefit transfer[2] because of their accuracy. However, time and resource limitation do not permit site specific studies in quick assessments. Benefit transfer is a compromise, in which findings of a previous study undertaken in a similar location (study site) is used for the concerned project site (policy site). Benefit transfer can potentially result in higher errors; therefore, its selective and cautious use is essential. The paper proposes original, site-specific studies and benefit transfer (or value transfer) methods at each stage of the IPA having carefully reviewed the available information and considering resource and time availability requirements for original studies.

The remainder of the paper is organized in the following way. Section II reviews the four–step IPA. At the end of section II, summary guidelines are provided for practitioners. Section III illustrates the application of the IPA using a hypothetical coal power plant in India. To evaluate uncertainty considerations, sensitivity analysis has been performed on the partial effect of relevant parameters. It should be noted that the methodology may also be applied to any type of power plant that burn fossil fuels such as diesel or natural gas. The methodology is also applicable for mobile source of air pollutants such as transport with necessary modifications to the dispersion modelling.

[2] Appendix 1 provides a brief discussion about benefit transfer method.

II. REVIEW OF METHODOLOGY

This study uses IPA in estimating the health costs of air pollution. The IPA is widely accepted and applied in many studies such as European Commission (2005) Krewitt, et al. (1999), Anderson et al. (1997), Daniels et al. (2000), Bhattarcharya et al. (2007), Chestnut, Ostro and Vichit-Vadakan (1997), Vichit-Vadakan, Vajanapoom and Ostro (2008), and Wong et al. (2006). The IPA framework allows for the evaluation of incremental pollution impacts arising from a project in a given location. The core of the methodology lies in the recognition of the simple causal chain that starts from the source of pollutant to the endpoints (the physical impacts) and their eventual valuation in monetary terms.

Estimation process of impact of air pollution on human health begins with identification of sources of pollution and their emission factors. Data on energy sources and their consumption levels with a given technology are used to estimate the emission concentrations near the source. This information is then combined with meteorological patterns to predict pollution levels at receptor sites. Once ambient concentrations in receptor sites are known, the incremental increase caused by the plant in focus is linked to incremental increase in mortality and morbidity. Finally, these increases are converted to monetary values. IPA methodology can be used to estimate health cost of air pollution from existing plants as well as plants under consideration for future implementation. When using this method, the analyst may be confronted with data limitation issues (e.g., inaccurate knowledge of meteorological conditions), failure to adequately account for long-range transport of pollutants, and a lack of proper accounting of emissions from non-point sources (e.g., trash burning). The steps involved in the IPA are illustrated in Figure 3. In principle, IPA framework is applicable in evaluating not only the impact on human health, but also a wider range of environmental impacts such as global warming, damages to ecosystems, and biodiversity loss.

Figure 3: Impact Pathway Approach

Emission Source
Specification of Site and technology
e.g., kg/year of pollutant emission

↓

Dispersion
Assessment of Atmospheric dispersion
e.g., increase in pollutant concentration in affected region (measured in $\mu g/m^3$)

↓

Physical Impact
Estimation of Health Impact
e.g., increased incidence of asthma due to particulate matter increase

↓

Cost of Welfare Impact
Valuation of Health Impact
e.g., cost of lowering risk of premature death

Source: Adapted from European Commission (2005).

A. Source Specification and Estimating Emissions

The first step entails collection of data such as fuel type, the quality of fuel, combustion characteristics, chemical composition of fuel, number of operation hours, and level of pollution control at the plant to quantify emissions from the plant under consideration. The primary output that is estimated at the end of this step is the emission rate of the various pollutants whose impact is being studied. The information required for estimating the emissions at the source (e.g., capacity, plant load factor, specific fuel consumption) is usually available or can be assumed. An example of this information is available in Table 5 of Section III.

B. Dispersion Modelling

Pollutants from the plant which are dispersed in the atmosphere cause changes in the pollution levels in the receptor sites. This second step quantifies the ambient pollution concentrations at the receptor site using a dispersion model. Computation-wise, dispersion modeling is the most involved step in IPA. Dispersion models enable calculating the ambient concentration increase of the pollutants being examined, at a location within the local area of the emission source (< 50 km) or in the region of influence (> 50 km). When a vertical mixing of the various pollutants in the atmospheric mixing layer occurs and secondary pollutants (e.g., sulphates, ozone) derived from primary pollutants also present, a carefully defined region of influence should be considered.

A mathematical simulation of air pollutants dispersion in the ambient atmosphere is carried out by the dispersion model. Dispersion models thus, predict downwind concentration of air pollutants emitted from emission sources (stationary plants in our case). For convenience, dispersion model types can be divided broadly into two: steady-state Gaussian-plume models and advanced models. From a practical standpoint, the greatest difference between model types is in the requirements of meteorological information and computer resources. However, some 'steady-state' models are highly sophisticated and not necessarily 'Gaussian', so there is very little to no apparent distinction at all (NIWA 2004). Gaussian-plume models, which are based on mathematical approximation of plume behavior, are the easiest models to use, primarily because assumptions applied in this model are quite simplistic. It should be noted that despite having simplistic assumptions, this type of model can provide reasonable results when used with necessary prudence and accurate data.

One of the key elements of an effective dispersion modeling study is to choose an appropriate model to match the scale of impact and complexity of pollution dispersion. When choosing the most appropriate model, the principal issues to consider are the complexity of dispersion (e.g., terrain and meteorology effects), and the scale and significance of potential effects including sensitivity of the receiving environment (e.g., human health versus effects on buildings). There are many proprietary and publicly available models that implement the two broad categories of models.[3] Steady-state Gaussian models have favorable characteristics that make them useful and convenient for estimating changes in pollutant load: these models require minimal computational requirements (i.e., they can be run on almost any desktop computer); and they require simple meteorological data (i.e., data set can be developed from standard meteorological recordings). Commercially developed data sets are often readily available for steady-state Gaussian models.

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[3] Gaussian-plume models: AUSPLUME, ISCST3 (EPA), AERMOD (EPA) and CTDMPLUS and advanced models such as CALPUFF and The Air Pollution Model (TAPM).

Simple excel based Gaussian-plume models can produce reasonably reliable results and can simulate ambient concentration variations within a zone that has medium-complex atmospheric and topographical conditions (not steep) with uniform spatial meteorology. It is also desirable that there are few periods of calm or light winds. A careful choice of Gaussian-plume model is required if the effects of deposition, chemistry or fumigation are to be simulated. Closed form solutions (i.e., an expression which can be substituted with suitable known values) are available for simple plume dispersion scenarios. The concentration variation in the local area can be described by a Gaussian plume dispersion model as given below (Latha and Shanmugham 2010):

$$C(x,y,z) = \frac{Q}{2*\pi*u*\sigma y*\sigma z} * \mathrm{e}^{\left(-\frac{y}{2*\sigma y^2}\right)} * (\mathrm{e}^{\left(\frac{-(z-h)^2}{2*\sigma z^2}\right)} + \mathrm{e}^{\left(\frac{-(z+h)^2}{2*\sigma z^2}\right)} \qquad \text{Eq. 1}$$

Where

- *C(x,y,z)* is the concentration of the emission (in µg/m^3) at any point *x* meters downwind of the source, *y* meters laterally from the center line of the plume, and *z* meters above ground
- *Q* is the quantity or mass of the emission (in grams) per unit of time (seconds)
- *u* is the wind speed (in meters per second)
- *h* is the effective height of the source above ground level (in meters)
- σ_y and σ_z and are the standard deviations of a statistically normal plume in the lateral and vertical dimensions, respectively. They are given by
- σ_y = (x *α/√ (1+0.0001*x), where α takes various values depending on stability conditions assumed
- σ_z = f(x), where f(x) takes various forms depending on stability conditions assumed[4]
- Stability conditions classifications – very unstable, moderately unstable, slightly unstable, neutral, somewhat stable and stable

In the subsequent case study illustration presented at the end, an Excel based Gaussian–plume model is used to evaluate the dispersion of pollutants from a point source. The model is sensitive to wind speeds used and atmospheric conditions. Assumptions of neutral stability have been used for illustrative purposes. More advanced models, based on availability of data and other resources, can be used but they are not within the scope of this study.

C. Measuring Health Impacts

Once the ambient concentrations of pollutants are determined through dispersion modeling, this information can be used to estimate the incremental increases in health impacts. Dose-response functions (DRFs) quantify the relationship between air pollution and health impacts.[5] A growing body of empirical evidence has demonstrated reasonably consistent and strong relationships between exposures to air pollution and a number of health effects. These analyses have highlighted the role of particulates as a principal mediator of toxic effects on health, especially on the cardio-respiratory system (Cropper et al. 1997; Anderson et al. 1997; Daniels et al. 2000; Arden et al. 2002; Wong et al. 2006; Patankar & Trivedi 2011). To better understand these relationships, Table 1 highlights cause–specific mortality and morbidity endpoints associated with some of the most prominent air pollutants.[6]

[4] Details of equation 1 are explained in Appendix 5.
[5] Appendix 2 provides a technical discussion of DRFs.
[6] An endpoint refers to a specific health-related incidence that arises as a result of exposure to the pollutant.

Most of the studies on this subject are conducted in the United States and other developed countries, as these analyses are computationally demanding and costly. Often transferring DRFs from developed countries have been practiced because of the difficulties faced in undertaking original studies in developing countries.

Table 1: Health Endpoints Associated with Air Pollutants

Outcome	Disease	Pollutant
Mortality	Respiratory disease Cardiovascular disease COPD Cerebrovascular event Ischemic heart disease Respiratory cancer	Particulate matter (TSP, PM_{10}, $PM_{2.5}$) NO_2 SO_2 CO O_3 (formed from SO_x and VOC)
Respiratory Hospital Admissions (RHA)	Respiratory disease Asthma COPD Cardiovascular disease Cerebrovascular event Congestive heart failure Acute and Chronic bronchitis Cough in children Lower respiratory symptoms	Particulate matter (TSP, PM_{10}, $PM_{2.5}$) NO_2 SO_2 CO O_3 (formed from SO_x and VOC)
Restricted Activity Days (RAD)		PM

Sources: *European Commission, 2005; Health Effects Institute (HEI), 2010.*

COPD = Chronic-obstructive pulmonary disease, VOC = volatile organic compounds.

While general tendency is to transfer available DRF results from developed countries to developing countries, this practice necessitates caution to avoid generating misleading results with possibly important implications on policy decisions. Pollution levels are markedly different between developed and developing country contexts, which may render extrapolated results invalid especially in cases where nonlinear response patterns exist. Another reason is that differences in health status, lifestyle, and socioeconomic circumstances (e.g., costs of avoidance behavior) between these two contexts tend to alter one's susceptibility to air pollution (Cropper et al. 1997). It is encouraging to know, however, that there has now been some progress in the Asian literature on this subject. One good example showing a systematic progress in this area is the Health Effects Institute Reports (2004 and 2010), wherein a wealth of Asian studies has been quantitatively summarized for the short-term exposure effects of air pollution.

In cases where primary calculations of DRF estimates are not computationally feasible, the needs arise for transfer of DRF functions in project assessments covering Asian populations. Appendix 1 provides a brief description about the benefit transfer. A south–south transfer (i.e., developing country to another developing country transfer) of DRF may be a more suited approach rather than north-south transfer (Developed country to developing country transfer) because south–south transfers allow the use of studies that were conducted in similar socioeconomic and physical environmental setting. This would reduce errors in the transfer of DRF from a study site (where original detailed DRF estimation was undertaken) to the policy site (where quick assessment of health cost of air pollution is undertaken).

We reviewed the summary estimates presented in the HEI (2010) and examined whether they can be applied with reasonable reliability in DRF transfer for Asian countries. To properly address applicability concerns, we assessed how the original 82 studies used by HEI are selected, and examined whether they are representative of regional characteristics. We have also addressed reliability concerns by first checking whether the signs and magnitude of the HEI parameter estimates conform to theoretical expectations, and then, by comparing whether the estimates are broadly consistent with other studies from other regions. After a careful review, we are confident that these are the best available estimates to date in the Asian literature and that the combined analysis from these 82 studies is applicable to Asian countries. Appendix 2 provides a detail discussion of this review.

Table 2 presents HEI summary estimates for mortality health endpoints. The HEI summary estimates for mortality are fairly robust and broadly in line with theoretical expectations. Average estimates for morbidity endpoints are reported in Table 3. Unlike the HEI summary estimates that are reported for mortality, there are no actual quantitative summarizations reported for the morbidity endpoints except for respiratory hospital admissions (RHA). This is because HEI summary estimates are only calculated when four or more studies provided estimates for individual pollutant-outcome pairs. Hence for morbidity endpoints, we have calculated the point estimates using a simple average from a range of individual estimates presented in the HEI Report.

Table 2: Summary of Estimates of Percent Change in Mortality Outcomes

Pollutant / Mortality causes	Percent Change[a] (Point estimate)	Percent Change (95% CI)
PM_{10}		
All causes, all ages	0.27	0.12 to 0.42
All causes, >= 65	0.45	0.29 to 0.61
Respiratory, all ages	0.86	0.34 to 1.39
Respiratory, >= 65	1.09	0.55 to 1.63
Cardiovascular, all ages	0.36	0.09 to 0.62
Cardiovascular, >= 65	0.53	0.53 to 0.75
NO_2		
All causes, all ages	0.98	0.54 to 1.42
Respiratory, all ages	1.74	0.85 to 2.63
Cardiovascular, all ages	1.08	0.59 to 1.56
So_2		
All causes, all ages	0.68	0.40 to 0.95
Respiratory, all ages	1.00	0.60 to 1.40
Cardiovascular, all ages	0.95	0.30 to 1.60
COPD, all ages	1.72	0.10 to 3.36

Source: HEI. 2010.

Note: [a] Per 10µg/m^3 increase in ambient pollutant concentration.

Table 3: Estimates of Percent Change in Morbidity Outcomes [a]

Pollutant / Health endpoint	Percent Change (Simple Average)	Percent Change (Range of Estimates)
PM_{10}		
RHA (All Respiratory Causes)	1.3	0.2 to 2.2
RHA (Asthma Incidents)	1.2	0.5 to 1.6
RHA (COPD)	1.5	1.1 to 1.9
CVHA (All CVD)	0.7	0.6 to 0.8
CVHA (Angina / Ischemic)	0.7	-
Loss of work days	31.5 days/1000 adults/year	29 to 39 days/1000 adults/year
NO_2		
RHA (All Respiratory Causes)	0.92	0.17 to 1.68
CVHA (Angina / Ischemic)	0.8	0.0 to 1.2
SO_2		
RHA	0.51	-0.17 to 1.19
RHA (Asthma Incidents)	1.4	1.2 to 1.7

Source: *Various individual studies reported in the HEI* (2010).

Notes: [a]Per 10μg/m^3 increase in ambient pollutant concentration, CVHA = cardiovascular hospital admissions, CVD = cardiovascular disease.

Taking into account the applicability and reliability considerations, we recommend the application of HEI summary estimates for mortality endpoints and the average estimates reported for morbidity in quick project assessments. Although these are the best available estimates in the Asian literature to date, we also recognize the fact that there may be some degree of uncertainty on the transferability of results especially on the estimates for morbidity endpoints. Therefore, further research in this area is desirable and recommended. Given the recent progress on this subject, we have good reasons to expect that more 'first-hand' analyses will become available from other Asian regions, which in turn results to combined analyses with more balanced representation and better accuracy on morbidity estimates. The average morbidity outcomes should be updated to increase accuracy of quick assessments of health costs of air pollution when new studies become available.

D. Economic Valuation

Once the health impact of increased pollution concentration is known with reasonable certainty, the fourth step of the IPA necessitates the conversion of these physical impacts to monetary values. This can be done by estimating the compensating variation necessary to maintain the same level of well-being after an increase in the risk of health effects associated with pollution exposure.[7] There are different economic valuation methods which can be used depending on the type of health outcome to be valued. In this section, we first discuss the available valuation approaches for premature mortality. Appendix 3 presents a discussion on valuation of mortality.

Valuation of Mortality

The process of valuing an untimely death is not an easy task by any means. Widespread confusion arises due to misunderstanding that valuation attempts to assign a value for human life. This valuation has a limited scope as it attempts only to estimate the willingness to pay (WTP) for small reductions in one's risks of dying as a result of an increase in pollution exposure. Clearly, economic valuation does not seek to measure the value of life itself but only the value of lowering the odds of premature death,

[7] Compensating variation is a welfare measure derived using Hicksian demand function. For a simple explanation, see Gunatilake (2003).

which should be equal to the sum of the amounts that affected individuals are willing to pay to reduce small risk of premature death.

The monetized value of avoided premature mortality can be estimated using the Human Capital Approach (HCA) and the Value of Statistical Life (VSL). The HCA is a productivity-based approach, which presumes that the social worth of an individual is a function of his market productivity. This is a useful starting point, but for most practical purposes HCA may not be sufficient. A few major limitations of HCA are worth noting; HCA is affected by discriminations in wage setting, (i.e., wage based on a worker's gender and race) and labor market imperfections. Thus, the valuation resulting from the use of the HCA can at best be viewed as a lower bound estimate. The VSL on the one hand better aligned with economic theory, which can be estimated using either a stated preference survey (e.g., contingent valuation or CV) or derived from labor market data (e.g., hedonic wage). Wage-hedonics analysis has been particularly appealing for regulatory agencies as its application is more straightforward using readily available market data (See Appendix 3 for detailed discussion of VSL).

We have examined some recent valuation studies in order to have a better understanding of the possible range of VSL estimates in the US and developing economies (see Appendix Table 3.1 for details). Review of this literature points out that the most reasonable values are in the range of $5 million to $12 million, with a median estimate of $7 million derived from more than 60 hedonic labor market studies in US (Viscusi and Aldy 2003). This is in line with the mean wage-hedonics estimate of $7.4 million (in 2006 dollars), which is the US environment protection agency's (EPA) default guidance in the quantification of mortality risk reduction benefits for all risk contexts. A recent meta-analysis of 26 CV studies reports estimates between $0.13 million to $33.58 million (in 2004 PPP converted prices), with air pollution estimates coinciding with the lower bound of this range (Dekker et al. 2011).

Amongst the Asian studies, Vassanadumrongdee and Matsuoka (2005) have estimated mean VSL as $0.74 million to $1.32 million related to air pollution, while related to road safety as about $0.87 million to $1.48 million for Thailand. For PRC, Hammitt and Zhou (2006) have estimated the mean VSL as about $15,000 - $30,000 in Anquing, $45,000-$60,000 in Beijing, and $100,000-$180,000 in rural areas. The estimates for PRC are somewhat sensitive to modeling choices and location and should therefore be interpreted as lower bound estimates.[8] For India, Madheswaran (2007) estimates VSL as about Rs.15 million ($340,000), based on a sample of 1000 workers from Chennai and Mumbai. A general observation is that developing countries tend to have smaller estimates of VSL than those in developed countries. Aside from income and wealth, the VSL may depend on the characteristics of the individual (e.g., age, health, life expectancy), characteristics of risk (e.g., type of risk, latency between exposure to a hazard and resulting fatality) and the magnitude of risk reduction people have been asked to value, but the direction of such effects remains somewhat empirically and theoretically ambiguous (Lindhjem et al. 2012; Hammitt and Robinson, 2011; Alberini et al. 1997).

The VSL method is well-grounded in economic theory, but the general requirements for data collection and time may force the use of benefit transfer (BT), also known as value transfer (VT). The principal focus in this context is one in which previous valuation evidence (i.e., WTP) from other countries/regions is transferred to another site. The BT is used to transfer values of any kind and for

[8] The PRC estimates are 100 to 1,000 times smaller than the US estimates. One possible reason cited is that the mortality-risk reduction presented in the CV questions (1 or 2 per 1,000) is much larger than the risk reduction typically presented in CV studies (1 or 2 per 10,000). It should be noted that respondents are somewhat indifferent with the magnitude level of risk reduction.

any health endpoint. Most of the applications of air pollution related to BT were undertaken to transfer values between two countries, although this merits some caution when used as basis for policy-making (Alberini et al. 1997; Brouwer 2000). Earlier we discussed that several factors such as the differences in income and risk contexts affect VSL estimates. Hence, the key element to consider is the degree of similarity between the two sites, as this would determine the magnitude of the errors. The general recommendation is that when transferring across similar goods and sites, a direct transfer without much adjustment is likely to be sufficient (DEFRA 2009).

Overestimation or underestimation may arise in BT when risk contexts and other characteristics associated with the source studies and policy sites are not properly controlled for. Because of the differences in these factors, the VSL literature provides some guidance on how these differences can be moderated (See Appendix 1). When adjusting unit transfers between two countries, for instance, an elasticity of WTP with respect to income may be applied to correct income differentials. A range of reasonable estimates is available in the empirical literature, 0.4 to 0.6 (Alberini et al. 1997; Viscusi and Aldy 2003). Alternatively, the simplest approach is to assume an income elasticity of WTP that is 1.0 (Alberini et al. 1997). For adjusting unit transfers between two risk contexts, Dekker et al (2011) and Rowlatt et al (1998) have estimated a range of correction factors (1.80 to 2.0) for transfers between road safety to air pollution contexts.

It is worth noting that an active debate is on-going about the relative merits of BT in the empirical literature, given different levels of errors involved with health outcomes and risks to subpopulations. For example, Rozan (2004) has conducted a methodological exercise to test the reliability of BT by carrying out a simultaneous CV study under similar conditions on two neighboring sites (i.e., Strasbourg, France and Kehl, Germany). Results indicate high transfer error, which suggests that WTP significantly differs between the two sites. In particular, Kehl reported higher estimates of WTP than Strasbourg, implying stronger sensitivity to environmental problems in Germany. However, Rozan (2004) stresses that a transfer error rate of 15% or less may be acceptable if benefits are to be used for cost–benefit analysis. Relatedly, Hammitt and Robinson (2011) suggest that adjustments for the effects of income and latency may not be sufficient to support adjustments of VSL for an individual's underlying health status. In contrast, Alberini et al. (1997) have tested the validity of transferred values from the US versus direct WTP obtained by a CV study in Taipei,China, and they conclude that transfer between these economies are valid (using both adjusted unit transfers and value function), but remained cautious about interpretation of their results. Because of inconclusive nature of the on-going discussion, our general recommendation is to apply recently estimated VSL originating from a study site located in the same country with similar characteristics.

Valuation of Morbidity

Given the multiple health endpoints that characterize morbidity, valuation of morbidity impacts can be challenging. The actual WTP which captures the total welfare effect associated with the illness episode involves four key cost components (Rice 1966, Alberini and Krupnick 2000, and Champ et al. 2003):

(i) Cost of medical care associated with the treatment of an illness episode. This is a direct expenditure consisting of doctor's consultation fee, cost of medication and hospital admission, and cost of travel involved in seeking medical help;

(ii) Cost of productivity (income losses) as a result of work loss day (WLD) or restricted activity day (RAD). This reflects the opportunity cost of reduced productivity or output foregone because of an illness episode. This is an indirect expenditure but can be determined through the prevailing market driven wage rate;

(iii) Cost of averting actions to reduce exposure to pollution, is fundamentally based on the assumption that a rational person will take defensive actions as long as the value of the damage avoided exceeds the cost of defensive actions. The cost depends on the choices of defensive behaviors (e.g., wearing a smog mask or personal air filter, deciding to stay indoors, closing the windows, or even temporarily moving out of the area). This component can be derived using WTP, either revealed through averting expenditure questions or inferred through CV questions; and

(iv) Cost of disutility resulting from restrictions imposed by the illness episode, i.e., discomfort, pain, and anxiety. This is the only component deemed to be non-market in nature, which can only be inferred through a WTP survey.

The first two of the four components– direct medical expenditures and income losses– are cumulatively known as cost-of-illness (COI). The COI is therefore expected to be a lower bound on WTP (Champ et al. 2003). The COI differs widely across settings, even to the most recent and geographically closest location from the policy site. Outpatient treatment and hospital admission costs, for instance, tend to be more expensive in developed countries compared those in to developing countries. It is therefore necessary to carry out a primary COI study to properly accommodate these constraints. However, just by doing so would result in an omission of the latter two components– the psychic costs of illness episode and behavioral adjustments that people make in order to reduce exposure to pollution (Champ et al. 2003; Chestnut et al. 1997).

As shown in few studies that have empirically compared WTP estimates with COI estimates, considerable variation tends to exist between WTP and COI figures (Alberini and Krupnick 2000; Chestnut et al. 1996; Dickie and Gerking 1991; Rowe and Chestnut 1985). In these studies, estimates of WTP can range from 1.48 to more than four times COI figures, implying that the value of disutility and cost of averting actions can be quite significant. It is not be conceptually valid to assume that the economic value of disutility is zero, it may be necessary to obtain WTP estimates in order to fully capture the total welfare effects. On the one hand, if one is to commission a primary valuation study such as a CV survey, it would be practically more cumbersome than COI (Chestnut et al. 1996).

Therefore, one realistic approach entails taking stock of practical considerations and at the same time, applying a theoretically consistent welfare measure for the valuation of morbidity. In this emphasis, we recommend applying a two-step valuation approach– the first step is to carry out a primary COI study to incorporate locally-specific considerations, and the second step is to apply a scaling factor to the COI estimates in order to derive WTP. This two-step valuation approach is recommended so that total economic effects are counted, as the direct application of COI values are likely to underestimate the total social costs of morbidity.

The scaling factor is computed using the average WTP/COI ratio of three studies that have queried individuals directly about their WTP to avoid illness, and for which have compared these WTP estimates with COI estimates incurred by the same subjects (Alberini and Krupnick 2000; Dickie and Gerking 1991; Rowe and Chestnut 1985). We derive 1.65 as lower bound and 2.16 as central scaling factor (Table 4). The estimated upper bound WTP/COI ratio of 3.14 is taken from the mean highest central values also reported by Alberini and Krupnick (2000), Dickie and Gerking (1991), and Rowe and Chestnut (1985). See Appendix Table 3.2 for details.

Table 4: Summary of WTP/COI Ratios of Pollution-related Health Outcomes [a]

Outcome	Risk context	Lower bound	Central value	Upper bound	Reference
Minor illnesses	PM	1.48	1.88	2.26	Alberini and Krupnick (2000)
CBR and CRF	Ozone	1.98	3.00	4.17	Dickie and Gerking (1991)
Chronic asthma incidence	Carbon monoxide	1.50	1.61	3.00	Rowe and Chestnut (1985)
Computed Average Scaling Factor	–	1,.65	2.16	3.14	–

Source: *Compiled by authors.*
Note: [a]CBR = chronic obstructive respiratory disease, CRF = compromised respiratory function

As shown in Table 4, WTP/COI ratios tend to vary depending on the risk context, severity of illness, pollution level, and model specification, among others. Alberini and Krupnick (2000) estimate the WTP/COI ratio between 1.48 at very low PM concentrations (25µg/m3) to 2.26 at highest PM readings (350µg/m3). This study has estimated the WTP for avoidance of minor health problems associated with air pollution using 1991-92 data in Taipei,China. Interestingly, Alberini and Krupnick (2000) find that such range is similar to that of other US studies despite differences in geographic conditions. The same is reflected in the empirical studies of Rowe and Chestnut (1985), wherein a WTP/COI ratio ranging from 1.5 to 3.0 has been estimated for patients with chronic asthma episodes. Whereas Dickie and Gerking (1991) have estimated this ratio to be between 1.98 to 4.17 for patients with COPD and compromise respiratory function, depending on the city, ozone concentration, and model specification used. General consensus suggests that an upper bound tends to characterize minor health effects (e.g., eye and throat irritation) and chronic respiratory illness (e.g., asthma), while the lower bound tends to represent major illnesses (e.g., angina or cancer) since a significant fraction of WTP is attributed to pecuniary costs (Chestnut et al. 1996; Rowe and Chestnut 1985). The overall mean scaling factor of 2.16 is therefore within the bounds established in these studies, and is only a preliminary representation of selected illnesses. Our approach is impaired by the small number of available primary studies and the lack of geographic diversity on this subject, and we provide a broad uncertainty bound to reflect this fact.[9]

E. Guidance to Practitioners: A Summary

Following the above discussion on review of methodology and research findings we provide the following recommendation for applying IPA for project related quick assessment of health cost of air pollution from power plants. Our recommendations focus as to where original studies should be undertaken or BT can be applied at different stages of the IPA.

Step 1: The first step pertains to specification of the emission source and details the technology used, the fuel type, quality of fuel, combustion characteristics, emission composition, and level of pollution abatement at the plant. This establishes the emissions profile which provides the constituents of the emission, concentration and emission rates. These specifications will feed into step 2. Primary data on step one should be collected for particular plant under evaluation.

[9] See Appendix 3.2 for summary of literature comparing WTP and COI estimates.

Step 2: The second step is the dispersion modelling where the spatial distribution of pollutants released from the point source is calculated. A suitable dispersion model should be selected from available models based on the nature of pollutant being studied and the availability of meteorological and topographical data. Location specific meteorological data should be used to predict the ambient concentrations of pollutants for the affected populations.

Step 3: The third step is the estimation of the health impact of the increased pollutant concentration in the air at the receptors. This is measured in terms of morbidity (various endpoints) and mortality increases through the use of a DRF. The physical impact is calculated as the product of the DRF, the calculated incremental pollutant concentration and the population impacted by the increase. The DRF information given in Tables 4 and 5 are recommended for estimating the health impacts in the Asian region.

Step 4: The final step is the monetization of the physical impact through a value associated with mortality and morbidity. We recommend the VSL as the most suitable method for assessing mortality costs. Undertaking a primary VSL for this type of study is not easy. The BT method may be used as a practical alternative. Practitioners, however, should look for most recent and geographically closest studies when applying BT method. Morbidity is valued using COI approach. The sum of mortality impacts and morbidity impacts (after monetization) yields the overall economic impact of health endpoints associated with increased pollutant load. If values specific to the area in focus are not available, it is recommended to conduct a COI study. Recommended WTP/COI figure in this paper are not based on adequate number of studies. Therefore it should be used with caution. If time and resource permits, it is recommended to undertake a CV study to estimate the WTP for reducing morbidity impacts.

The above described methodology is applied to illustrate estimation of health cost of a coal– fired power plant in India in the next section. The methodology can be applied to any type of power plant that burn fossil fuels such as diesel, natural gas, and furnace oil, among others. It can be used to estimate pollution costs on any stationary source such as industries too. The methodology is applicable for mobile source of air pollutants such as transport. However, dispersion modelling needs major revision in case of mobile pollution sources.

III. AN ILLUSTRATION

In this section we illustrate the application of the methodology proposed in the Section II. We apply the methodology to a coal power plant to illustrate the four steps of IPA. Additionally, a sensitivity analysis has been performed to examine the effect of relevant parameters such as different abatement levels at plant site, DRF, scaling factor (WTP/COI ratio), and hospital costs (private or public) on the base case results (See Appendix 6). It is important, however, to recall that valuation of health impacts as a component of the project preparatory economic analysis is generally conducted under time and resource constraints. Hence, establishing clarity on the purpose of these types of assessments is necessary. As Curtiss and Rabl (1996) explains;

"A problem arises from the fact that air pollution damage depends on the sites of emission and receptors, whereas from the point of view of policy it is not practical to try and take into account each and every local detail. Rather one needs guidelines that are a compromise between precision and practicality"

It is in this spirit the valuation exercise below is presented. While compromising on certain details of the valuation process, care is taken to ensure that reasonable accuracy is maintained for the following parameters: pollution concentration at site, pollutant dispersion and ambient concentration, DRF, and economic values. This illustration uses one of the chosen sites for a super critical coal power plant proposed by the government of India, which will be located in Cheyyur in Kancheepuram district in of Tamil Nadu. Since the plant is still in the preliminary planning stage, we apply representative parameters for technology used, efficiency, among others, which reflect the conditions of operation of another super critical coal power plant located in Mundra, Gujarat. In line with the IPA methodology discussed in the previous section, the details of the four steps are provided below.

A. Technology and Pollutant Load

Only one 800 MW unit of a proposed 4000 MW plant has been considered. The assumed characteristics of the thermal power plant and fuel specification are outlined in Tables 5 and 6, respectively. The efficiencies assumed are in line with the highest standards of the industry and are representative of the future of thermal energy generation. The fuel with a calorific value of 4000 kcal/kg, from an indigenous source would have typical characteristics as indicated in Table 6.

Table 5: Pollution Load from 800 MW Coal Power Plant

Assumptions	
Capacity	800 MW
Plant Load Factor	85%
Specific fuel consumption	0.53 kg/kWh
Fuel consumption rate	8664 Tons/day
Conversion Efficiency	90.0%
Cycle Efficiency	45.0%
Overall Efficiency	40.5%
Heat rate	2123.5 kcal/kWh
Unabated Pollution Load (tons/day)	
SO_2 production	139
NO_2 production	171
CO_2 production	10,801
PM_{10} production	358

CO_2= carbon dioxide; kcal=kilocalorie; kg= kilogram; kWh=kilowatt hour; MW= megawatt; NO_2= nitrogen dioxide; PM10= Particulate matter up to 10 micrometers in size; SO_2= sulfur dioxide;

Source: Authors' estimates.

The assumptions about calorific value, ash, and sulfur content are typical of Indian sub-bituminous coals. Imported coal would be more desirable, but a significantly more expensive option. The resulting emissions from the proposed unit are calculated based on simple stoichiometric ratios for the combustion products. Table 5 outlines the unabated pollution load at the plant site, the highest of which is CO2 production at 10,801 tons a day. The net emissions given a standard abatement process and technology are calculated based on efficiency assumptions of the abatement devices. Table 7 provides the pollution load after standard abatement.

Table 6: Fuel Specification

Coal composition	Percent (%)
C	34.0
S	0.8
N	0.6
Ash	35.0

C=carbon, N=nitrogen, S=sulfur

Source: Authors' estimates.

Table 7: Pollution Load at Plant Site

Emission	% Control	Emissions after control (tons/day)
SO_2	95.0	6.9
NO_2	85.0	25.6
PM_{10}	99.0	3.6

Source: Authors' estimates.

B. Dispersion Modelling

The figures corresponding to the emissions in Table 7 are used as inputs to the dispersion model. The parameters required to model the dispersion of pollutant are derived from the power plant at Mundra (Table 8). The locally relevant parameters such as wind speed and direction are approximated using the data available for a nearby station (i.e., Cuddalore is located 50 kms. away from Cheyyur). To simplify assumptions associated with the calculations, moderately unstable conditions were assumed in assigning the stability parameter.

Table 8: Dispersion Parameters

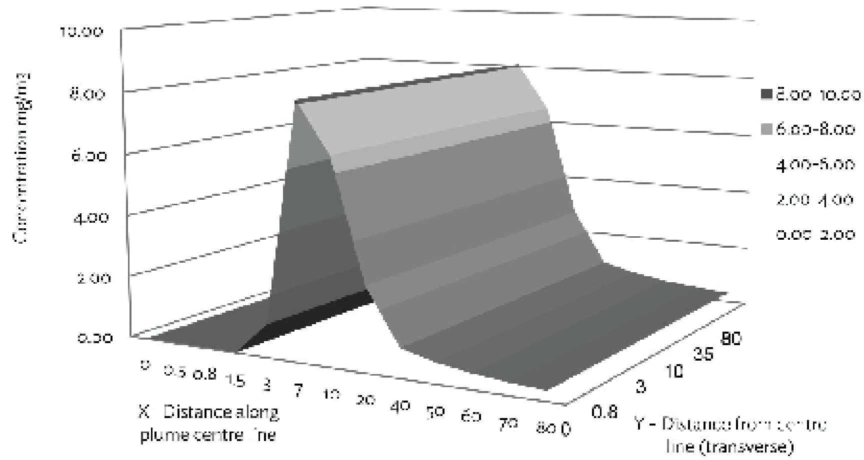
	Value
Stack height (m)	275
Stack diameter (m)	6
Gas exit velocity (m/s)	5
Gas exit temperature (C)	200
Ambient Temperature(C)	30
Wind Velocity (m/s)	1

NO2= nitrogen dioxide, PM10= Particulate matter up to 10 micrometers in size, SO2= sulfur dioxide.

Source: Authors' estimates.

A Gaussian plume model was evaluated using the parameters established in Table 8. The above and ground level concentration profile over a 100km radius was established (See Appendix 4 for a detailed information of the dispersion parameters). Figure 4 shows the variation of PM_{10} concentration.

Figure 4: Concentration Surface Profile of Pollutants Downstream of Source



Source: *Authors estimates.*

Note: Output from the Gaussian-Plume model for PM_{10} dispersion.

Since the concentration profile (at the ground level) is a continuous variable rather than discrete one, the increased pollutant load was evaluated in specific urban clusters around the power plant. A list of all major towns and cities within a 100 km radius was initially considered and only the significant population centers were retained for this exercise (Table 9). The estimated marginal increases in pollutant concentration (mg/m3) for SO2, NO2, and PM10 are computed using the pollution load data and parameter values discussed above for the coal power plant project.

Table 9: Pollutant Concentration at Discrete Population Cluster Locations

City	Population	Distance from Cheyyur (kms)	SO_2 Increase per 10 ug/m^3	NO_2 Increase per 10 ug/m^3	PM_{10} Increase per 10 ug/m^3
Chennai	6,540,462	85	0.57	2.10	0.29
Pondicherry	505,959	51	1.12	4.16	0.58
Kancheepuram	188,733	64	0.86	3.19	0.45
Cuddalore	158,634	75	0.69	2.55	0.36
Thiruvannamalai	130,567	100	0.44	1.62	0.23
Arakonam	78,686	89	0.57	2.10	0.29
Thindivanam	67,737	40	1.55	5.72	0.80
Chengalpattu	62,852	39	1.55	5.72	0.80
Arani	60,815	85	0.57	2.10	0.29
Thiruvallur	45,732	89	0.57	2.10	0.29
Melvisharam	36,757	100	0.44	1.62	0.23
Sriperumbudur	16,156	69	0.86	3.19	0.45
Ananthapuram	6,138	73	0.69	2.55	0.36
Vandavasi	29,620	52	1.12	4.16	0.58

Source: *Authors' estimates.*

C. Quantification of Health Impacts

After estimating the incremental increase in pollutant concentration, the next step entails quantifying the physical health impacts. We use the mean estimates of DRFs reported for both mortality and morbidity outcomes presented in Tables 2 and 3. Here we apply the direct transfer of average values in these tables.

Mortality Impacts

Baseline deaths associated with acute mortality are first computed based on national health figures (See Appendix 4). After applying the estimated increase in pollutant concentration from the dispersion modelling and the mean risk rate reported in Table 2, we estimate the total increase in mortality incidence at about 94 premature deaths per year (Table 10). It should be noted that the relatively small increase in premature mortality incidence may be attributed, to a large extent, to the high abatement measures being applied at the plant site, removing about 95%, 85%, and 99% of SO_2, NO_2, and PM_{10} respectively. In the absence of any abatement, mortality incidence is estimated at 1,243 premature deaths per year.

Table 10: Estimated Increase in Mortality Incidence Associated with Air Pollution, by City

City	Population	Distance from Source (kms)	SO_2	NO_2	PM_{10}	Total by City
Chennai	6,540,462	85	10	56	3	70
Pondicherry	505,959	51	2	9	1	11
Kancheepuram	188,733	64	0	2	0	3
Cuddalore	158,634	75	0	2	0	2
Thiruvannamalai	130,567	100	0	1	0	1
Arakonam	78,686	89	0	1	0	1
Thindivanam	67,737	40	0	2	0	2
Chengalpattu	62,852	39	0	1	0	2
Arani	60,815	85	0	1	0	1
Thiruvallur	45,732	89	0	0	0	0
Melvisharam	36,757	100	0	0	0	0
Sriperumbudur	16,156	69	0	0	0	0
Ananthapuram	6,138	73	0	0	0	0
Vandavasi	29,620	52	0	1	0	1
TOTAL	7,928,848	–	14	75	5	94

Source: *Authors' estimates.*

Notes: The average DRFs are based on the mean HEI estimates reported in Table 2 of Section 2C. For SO_2 and NO_2, these estimates are based on all cause (exc. age and age-specific) values. For PM_{10}, these estimates are based on the cumulative values cause and age-specific (cardiovascular, respiratory, & >65).

Morbidity Impacts

Respiratory Hospital Admissions (RHAs) and Work Loss Days (WLDs) are two of the most common representations of morbidity incidence; hence these are reflected in this exercise. Using the same information for pollutant concentration, we have estimated the cases per person for a given increase in pollutant load (per 10µg/m^3). Table 11 reports an increase of approximately 22,862 in RHA (all-cause) and 8,117 lost work days annually because of illnesses linked to pollution from coal-fired power plants such as minor respiratory infections, cough, and asthma. Again, such increase in morbidity incidence is moderated by the high abatement level for SO_2, NO_2, and PM_{10} applied at the plant site. Without abatement measures, RHA incidence is estimated at approximately 500,000 cases and WLD at 800,000.

Table 11: Increased Morbidity Incidence by Pollutant type

City	Respiratory Hospital Admissions (all causes)				WLD
	SO_2	NO_2	PM_{10}	TOTAL	PM_{10}
Chennai	1,895	12,636	2,494	17,025	6,044
Pondicherry	290	1,935	382	2,607	926
Kancheepuram	83	555	109	747	265
Cuddalore	56	372	74	502	178
Thiruvannamalai	29	195	39	263	93
Arakonam	23	152	30	205	73
Thindivanam	53	357	70	480	171
Chengalpattu	50	331	65	446	158
Arani	18	117	23	158	56
Thiruvallur	13	88	17	119	42
Melvisharam	8	55	11	74	26
Sriperumbudur	7	47	9	64	23
Ananthapuram	2	14	3	19	7
Vandavasi	17	113	22	153	54
TOTAL	2,545	16,968	3,350	22,862	8,117

Source: Authors' estimates.

D. Economic Valuation

Following previous discussions in Section II D, health outcomes are monetized in two ways. For the valuation of acute mortality, we apply a benefit transfer of a VSL estimate from Madheswaran (2007). This local study estimates VSL at Rs. 15 million ($331, 858), based on a sample of 1000 workers from Chennai and Mumbai. The cumulative cost for an increase in mortality risks is estimated at $31.08 million (Table 12). Meanwhile morbidity outcomes are monetized using a two-step valuation approach. First, COI is computed for RHA using available cost information from Patankar and Trivedi (2011). This study was the most recent study available in the literature and it was conducted in Mumbai which has very similar social economic setting to the study site. The average daily wage rate from the Ministry of Labor and Employment, Government of India was used to value WLD. Because cost of hospitalization and medical care from public services is likely to be subsidized, the COI derived from this should only be considered as lower bound. Hence to accurately reflect real cost of hospitalization and medical care, we assume that total RHA cost constitutes about 75% private and 25% public treatment. The estimated COI is then multiplied to the mean scaling factor (2.16) to allow for the estimation of WTP.[10] As shown in Table 12, the WTP is estimated at $15.12 million. Cumulatively, the social cost of morbidity stands at about $46.21 million.

[10] The scaling factor (WTP/COI) is calculated from three studies (Alberini and Krupnick 2000; Dickie and Gerking 1991; and Rowe and Chestnut 1985).

Table 12: Valuation Estimates for Mortality and Morbidity

Item	Increased incidence	Valuation basis	Valuation estimate Rs.	$
Acute mortality	94	15,000,000	1,403,742,523	31,083,758
Morbidity (WTP)				
RHA	22,862	13,750	679,009,882	15,035,648
WLD	8,117	224	3,933,990	87,112
TOTAL	–	–	2,086,686,395	46,206,519

RHA = Respiratory Hospital Admission, WLD = Work loss days, WTP = Willingness to Pay

Source: Authors' estimates.

In Table 13, it can be observed that pollutant load is much lower after emission control. For instance, PM10 emission savings are estimated at 355 tons/day following a high level of abatement. On a per kWh (per unit) basis, the total cost imposed by local air pollution is computed to be about $1.05 cents. A summary of a review on per kWh health costs by Soderholm and Sundqvist (2006) is presented in Appendix 5. The only Indian estimate is this review is close to our estimate and it is within the range of estimates. This shows that our estimate is reasonably accurate. However, it must be noted that the estimated health cost achieved only because of the expenditures incurred in installing abatement measures (FGD and SCR) for the removal of pollutants to an assumed degree. In the absence of abatement measures, the hefty cost of pollution on society is estimated at $12.58 cents per kWh. Put differently, if the industry invests about $0.28 cents per kWh for pollution control spending (removing about 85% to 99% of three major pollutants), this brings down the health cost imposed on society to $1.05 per kWh with a net welfare gain of $11.25 cents per kWh.

Table 13: Pollution Load by Emission Control

Pollutant	Zero Pollution Control (tons/day)	With Pollution Control (tons/day)	Net gain
SO2	138.62	6.93	131.69
NO2	170.80	25.62	145.18
PM10	357.94	3.58	354.36
TC/kWh ($)	12.58	1.05	11.53

NO2= nitrogen dioxide, PM10= Particulate matter up to 10 micrometers in size, SO2= sulfur dioxide, TC/kWh= total cost per kilowatt hour.

Source: *Authors' estimates.*

E. Sensitivity Analysis

As discussed above, our base case results are built on several important assumptions ranging from the level of abatement to that of DRF and economic parameter values. First, the coal-power plant achieves high level of abatement for SO_2, NO_2, and PM_{10} at 95%, 85%, and 99% respectively. Second, we apply the mean HEI estimates for the quantification of mortality and morbidity outcomes. Third, we use a locally estimated VSL from Madheswaran 2007, estimated at Rs. 15,000,000, and apply benefit transfer for valuing reduced risk of premature mortality. Finally, for the valuation of morbidity we use the COI from Patankar and Trivedi (2011), which has been estimated to consist of a mix of public (25%) and private treatment (75%), and we apply a mean scaling factor (2.16) to the COI in order to estimate the WTP. In this section we extend our analysis to take into consideration uncertainty in the estimation of physical impacts and their valuation. Table 14 provides an overview of the sensitivity parameter values.

To examine how much total health costs change in response to a change in a given parameter value, simulated results are generated using partial sensitivity analysis.

Table 14: Parameters used for Sensitivity Analysis[11]

Factors	Default	Sensitivity Parameters
1. Level of Emission Abatement SO_2-NO_2-PM_{10}	High 95%-85%-99%	Low 24%-21%-25% Average 48%-43%-50%
2. Mortality DRF	Average	Lower bound Upper bound
3. Morbidity DRF	Average	Lower bound Upper bound
4. COI estimates for valuation of morbidity	RHA (25% public & 75% private) + WLD	RHA (100% public) + WLD RHA (50% public & 50% private) + WLD
5. Scaling factor (WTP/COI ratio) for valuation of morbidity	Average (2.16)	Lower bound (1.65) Upper bound (3.14)

COI = cost of illness, DRF = dose response function, NO2-= nitrogen dioxide, PM10 = particulate matter, RHA = respiratory hospital admission, SO2 = sulfur dioxide, WLD = work loss days, WTP = willingness to pay

Source: *Authors' illustration.*

We start with the partial sensitivity analysis on the abatement levels. If we allow abatement levels to vary while holding other variables constant, results indicate steep declines in public welfare as implied by an increase in total health costs. In low levels of pollution control for instance, where in abatement measures remove between 21%–25% of emissions, the total health costs are estimated at $574 million compared to $46.21 million when abatement levels are high (Table 15). If we assume an abatement level with average pollution reduction i.e., control measures remove only about 43%–50% of emissions, health costs are estimated at $398 million holding other factors constant. This sharp increase in health costs of air pollution when abatement level is low points out to an important policy issue – monitoring of pollution control in power plants. Most of the time power utilities install the pollution control equipment to meet the upfront industry requirements, however may not necessarily be in operation at all times (in light of reducing maintenance and operating costs). Strong pollution control monitoring systems in Asian countries are therefore imperative.

Table 15: Total Health Costs by Abatement Level

	Total Health Cost		
Abatement Level	**Rs. (Million)**	**$ (Million)**	**Variation from base (%)**
Low	23,178.10	513.24	1011%
Average	17,970.51	397.93	761%
High	2,086.69	46.21	–

Source: *Authors' estimates.*

Note: Base results are highlighted.

11 Partial analysis has been applied to analyze how much results change in response to a change in given parameter.

Details of the sensitivity analysis are given in Appendix 6 and the summary results are given in Table 16. As shown in the Table 16, the health cost of air pollution is sensitive to all the examined parameters. The highest sensitivity is reported for change in pollution load followed by the mortality DRF. Given the non-linear nature of the relationships, upper bound parameter values cause more sensitivity compared to lower bound parameters. Overall the high sensitivity of the health cost estimates to pollution load, DRF and other parameters indicates that the analyst should be careful when undertaking these assessments and should try to use the location specific parameters to the extent possible.

Table 16: Summary Results of Sensitivity Analysis

Sensitivity Parameter	Health Cost of Air Pollution $ million/ year		
	Lower Bound	**Base Case**	**Upper Bound**
DRF (mortality)	32.19 (30%)	46.21	60.15 (30%)
DRF (morbidity)	37.50 (19%)	46.21	59.20 (28%)
WTP to COI ratio	42.64 (08%)	46.21	53.07 (15%)
COI	38.83 (16%)	46.21	–

COI = cost of illness, DRF = dose response function, WTP = willingness to pay

Source: *Authors' estimates.*

Notes: The lower bound of COI means 100% public health care. Percent deviations from base case are in parentheses. Dash (–) indicates that base case for COI is in itself the upper bound (constituting about 75% private cost and 25% public health care).

IV. CONCLUSION

Electricity generation poses inadvertent consequences on public health, severity of which depends on the type of energy resource used and the efficiency of pollution control policies, among others. However, as this type of externality is not measured directly in the market, assigning monetary values for the damage is necessary for making sound policy decisions. As shown in the paper, estimating health cost of air pollution from power plants is quite cumbersome and costly. Having reviewed the relevant literature, paper proposes pragmatic approach for estimating the health cost of air pollution in Asian countries. The proposed streamlined methodology is then applied to an 800 MW coal power plant in India. One pertinent observation from the analysis is that abatement using current technologies is critical to minimize the damage costs of thermal power plants. Results suggest that pollution abatement is economically efficient– with pollution control spending of $0.28 cents per kWh a net gain of $11.25 cents per kWh of avoided health cost can be achieved. Thus, significant public health benefit can be achieved by pollution abatement. Strengthening the available regulatory measures of pollution control and implementing a rigorous monitoring program can be justified based on the cost effectiveness of pollution abatement from coal power plants.

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APPENDIX 1: BENEFITS TRANSFER

This appendix briefly discusses issue of benefit transfer (BT) following the guideline set out by DEFRA (2009). Generally, BT is seen as a practical way of applying valuation evidence from previously undertaken studies (study site) to another setting (policy site).[12] For example if the underlying proposed change of the good in question is improvement in air quality and the analyst is evaluating the benefits of an air quality strategy or regulation for which the primary benefit is X premature deaths avoided, then there is a need to identify source studies with an identical or nearly similar context, of which estimated values of a premature death avoided have been calculated. The benefit (VSL) will then be transferred to the policy site in one of two ways: i) unit value transfer; ii) benefit function transfer.

1.1 Unit value transfer

This is the simplest of the two approaches, as it involves a straightforward pooling of data from all relevant primary studies and applying these estimates to the policy site. Although it is the easiest approach, this requires practical validation of the following components:

- Source valuation studies must be of high quality and based on adequate data. This constitutes a survey design that is consistent with economic theory and an econometric model with a theoretically consistent utility specification.
- Risk contexts (e.g., road safety; air pollution) must be similar. Good to be valued must be identical both in terms of the nature of the good and its provision change in both quantity and quality terms. For example in the case of a policy to reduce air pollution in urban areas, ideally the status quo and post-change levels of air quality and the levels of pollution concentration at the study site should match those of the policy site from which valuation evidence is to be transferred.
- Physical contexts and intra-temporal dimension such as distance to populations, characteristics of that population (e.g., sensitivity to environmental problems of the population), air quality index, and characteristics of pollution source (e.g., traffic or industrial activities) must be identical or nearly similar.
- When study site and policy site are not within the same country, monetary unit (i.e., country currency) remains unchanged and the transfer consists only in a multiplication of the mean WTP from the study site by the size of the affected population from the policy site.
- When study site and policy site are not within the same country, WTP of the policy site should be adjusted by considering the income difference in the two countries.

For example, if the VSL for the country X is available (D_X), the VSL for the country Y (D_Y) can be estimated as follows:

$$D_Y = D_X * (PPPGNP_Y / PPPGNP_X)^E \qquad \text{Eq. 1}$$

[12] Study site is where the valuation evidence is drawn, while the policy site is where the valuation evidence is to be transferred.

where $PPPGNP_X$ is the real gross national product per capita in purchasing power parity terms for country X and similarly for Y. The important parameter to note here is the income elasticity E of WTP. The rationale behind the formula indicated above is intuitive. The value in country Y needs to be scaled by the magnitude of the purchasing power differential between the two countries and in a manner which reflects local income elasticities. Some studies use an income elasticity of 1, and also use the real GNP values as opposed to values adjusted for PPP. There is no conclusive evidence available of the correctness of either approach. The ratio of PPP GNP (per capita) of the US to that of India in 2010 was around 15[13]. It should be noted that a benefit transfer from US EPA study using the above formula (assuming E = 1) would yield a VSL for India is about $460,000. In comparison to Madheswaran (2007), this number is different by about 26% and that can significantly alter the outcome of a project evaluation.

In some cases, mean unit values may be adjusted before transfer for studies concerning meta-analysis. An adjustment is also generally required to address differences across time (e.g., when study site value is old) or physical contexts (e.g., quality of the environmental good is not the same). An adjustment may also be necessary when risk contexts between the two sites differ. Dekker et al (2011), for example, suggests that an appropriate correction factor of 1.8 can be applied when transferring valuation evidence from road safety context of the study site to air pollution context of the policy site.

1.2 Benefit function transfer

This much more sophisticated approach uses the functional form and parameter estimates in the source study in order to derive a new set of value estimates specific to the policy site. As is the case with unit transfer approach, this necessitates that source valuation studies are based on adequate data, with sound economic methods and theoretically consistent empirical methods. Compared to unit value transfer, this approach is more flexible as it allows an adjustment on the characteristics of the population (e.g., age, sex, income, etc.) while keeping the coefficient estimates constant. The underlying principle is that the different explanatory variables of the WTP are similar in the study and policy good contexts, and one that is focused on the general relationships that economic theory suggests should hold across the study and policy good contexts.

To ensure that the results of the VT are defensible, the commodity valued and physical contexts have to be identical or sufficiently similar between the two sites. The general recommendation is that when transferring similar goods and sites, unit value transfer is likely to be sufficient. When transferring across similar goods, but dissimilar sites, value function transfer may be more appropriate and the specification of those functions should be restricted to include only generic variables for which there are prior economic expectations. Analysts applying the BT techniques are referred to the February 2010 edition of *Valuing Environmental Impacts: Practical Guidelines for the Use of Value Transfer in Policy and Project Appraisal* (DEFRA 2009) for a thorough discussion of the advantages and limitations of BT.

[13] Author estimates based on price of a basket of goods.

APPENDIX 2: QUANTIFICATION OF PHYSICAL IMPACTS

This appendix provides a brief discussion of the Step 3 of the IPA, where in we describe the statistical relationship between short-term exposures on air pollutants and their associated health effects. It is divided into three sections: Section 2.1 presents a technical discussion on Dose-Response Functions (DRFs) as one method of numerically examining such relationships; Section 2.2 presents a summary of Asian DRF studies. It should be noted that although we refer only to the 'dose-response' assessment, the same principle generally applies for studies describing the 'concentration-response' relationship.

Section 2.3 provides a qualitative assessment review of the HEI Report 2010 on the basis of applicability and reliability considerations, to augment our previous discussions on the application of a south-south transfer, and more specifically the use of HEI estimates for mortality and morbidity endpoints for quick project assessments requiring extrapolated results in Asian setting.

2.1 Dose-Response Functions

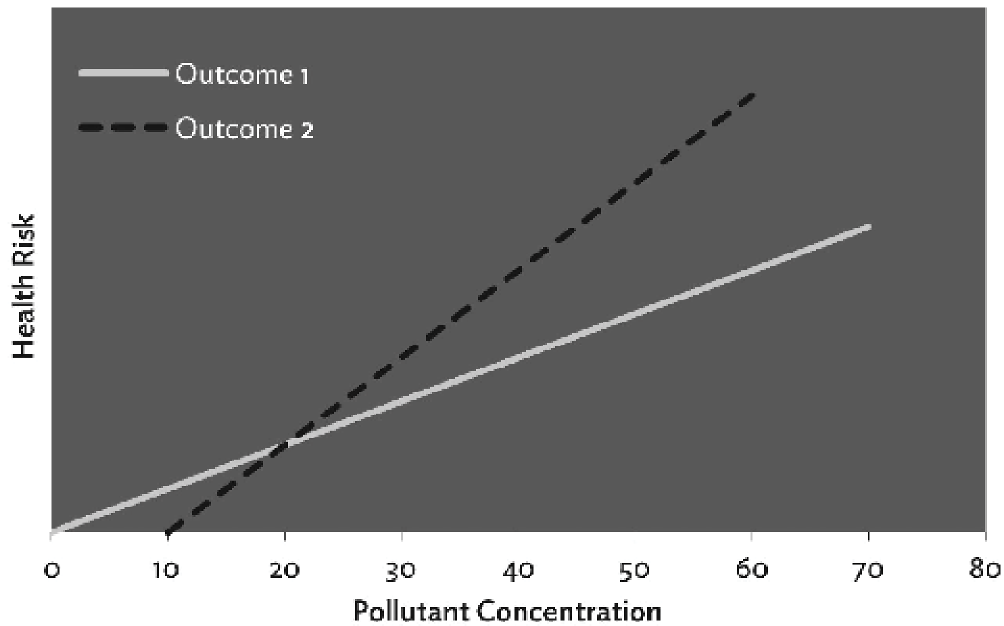
Recalling that in Step 2 we have estimated the air quality concentration from point emission sources using dispersion modeling, we now use these values to determine the physical health impacts resulting from a marginal change in the ambient air quality concentrations.[14] Once generated, the DRF allows us to estimate the likelihood and severity of health effects (the responses) occurring after an exposure to a specific amount and condition of pollutant (the dose) (US EPA 2012).

Intuitively, we can expect variations in response patterns– in that, aspects of this relationship may differ both in terms of the dose at which a response begins to appear and the rate at which it increases. These variations in response patterns can be attributed to a combination of several factors including differences in exposure routes, pollutant mix, and demographic structure, as well as the interaction between these factors (Cropper et al. 1997; HEI 2010). Since the magnitude of pollutant effect is a function of the dose, we can expect that the magnitude of response increases as dose rises. In some cases, there is a threshold dose present where in no response may be expected at lower dose levels.

As a simple illustration, Appendix Figure 2.1 consists of two possible associations. First, the unbroken line illustrates a case with no threshold, in which a response begins to appear once pollution concentrations are greater than zero. Second, the unbroken line illustrates a case where there is no response between zero and 10 concentrations, but beyond 10 the health risk rises more steeply, suggesting the presence of a threshold. Dose-response curves are often semi-logarithmic, i.e., amount of dose is plotted as the log of pollutant concentration, giving them their familiar 'S' or sigmoid curve (linear curve is shown only for illustration purpose). This simple illustration is useful to develop basic dose-response relationships. However, more complex relationships exist for many pollutants depending on the target organ and specified exposure.

[14] Point emission sources such as coal-burning power plants.

Appendix Figure 2.1: Dose-Response Functions



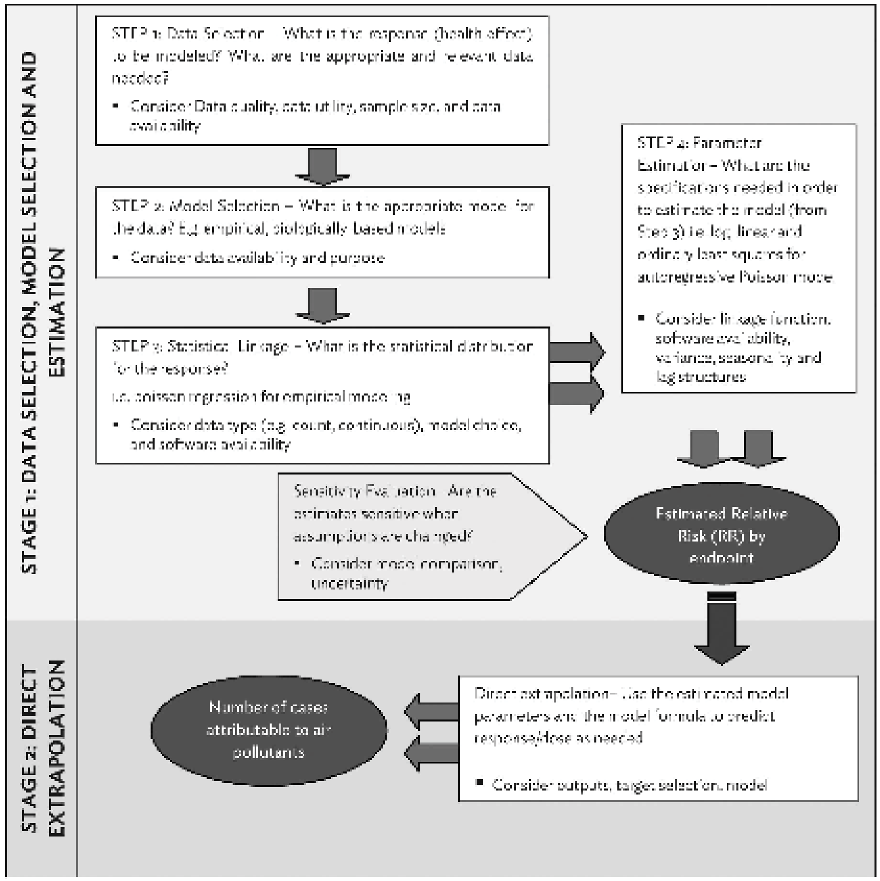
Source: Authors' illustrations.

Note: Actual shape of curve may be sigmoid ('S' shape).

In describing the basic dose-response assessment, we have adopted and modified the six steps which have been described in the IPCS (2009). Appendix Figure 2.2 provides a modified schematic overview of the basic dose-response assessment. DRF assessment is carried out in two stages: i) estimation of DRF; and ii) extrapolation. Estimation of DFR involve assessment of all data that are available or can be collected through experiments, selection of suitable econometric model, determination of statistical linkage and estimation of the model. For each of the included health endpoints, the output calculated is an estimate of the Relative Risk (RR), which can be interpreted as the increase in the probability of a given health effect associated with a given increase in exposure (e.g., 10 microns for particulates). After estimating the RR for every relevant endpoint, a sensitivity analysis is conducted. This exercise is crucial as it assesses whether the magnitude and signs of pollution parameters and their standard errors are not significantly affected. Once the RRs are estimated they will be used to predict the number of health endpoints for a given increase in air pollution. A more detailed discussion on the use of Dose Response Modeling can be found at the IPCS Report (2009).

As a case illustration to describe the first stage (Steps 1 to 4), we use the empirical analysis of Cropper et al. (1997) on daily mortality associated with exposure to particulates in Delhi. For Step 1, the study has aimed at examining health damages associated with air pollution in developing countries, and in particular, the effects of particulate matter on daily deaths in Delhi. Clearly, the data needed for this study include pollution variables (e.g., daily data on air pollution levels collected at relevant monitoring stations in Delhi); time-series mortality data for the period 1991-94 from New Delhi Municipal Committee, e.g., daily counts of non-trauma deaths by selected causes such as respiratory illness and cardiovascular disease as well as by age group; and control variables such as weather changes (e.g., average daily temperature, rainfall, and visual range), among others.

Appendix Figure 2.2: Basic Steps for Dose-Response Modeling Assessment



Source: Authors' illustrations.

For Step 2, the available data would require an empirical estimation in order to analyze the collected information. For Step 3, Cropper et al (1997) have employed a poisson regression model since daily mortality data are counts of rare events, and a log-likelihood function is necessary for the model to run (Step 4).[15] As these are time-series data, the model needs to be controlled from the seasonality in the data, e.g., time trend, dummy variables to account for the study year to account for population increases and other unobserved factors. After the estimates of RRs have been calculated, sensitivity analyses are conducted which may include adding another pollutant in the base model, correcting for over dispersion, or using monthly dummies to control for seasonality.

Measured in terms of a change in relative risk of a health endpoint per unit population (1µg/m^3 or 10µg/m^3), the DRF value represents the change in the number of health endpoints from a marginal change in the ambient air quality. DRF is commonly reported as a number of increased cases per unit population per unit increase in pollutant concentration. For example, in a time-series study on air pollution and mortality conducted by Rajarathnam et al. (2010) in Delhi, the results generate a DRF value of 0.15% for concentration– this can be interpreted as the daily mean for all-natural-cause mortality in all age groups increased by 0.15% as the mean PM_{10} concentration increased by 10µg/m^3.

[15] Other DRFs can be exponential.

The second stage covers extrapolation of estimated RRs. This would normally involve extrapolation to estimate the risk for exposures beyond the range of available observed data. This is necessary in order to make inferences about the critical region where the dose level begins to cause the adverse effect in the human population. For assessments based on human studies, it is a downward extrapolation of different exposure levels, as well as of different life stages (e.g., child) or different populations with different environmental factors that might affect exposure. There is perhaps more concern with transferring results to human studies when dose-response assessments are based on laboratory testing of animal subjects. This is because animal subjects are often exposed to higher doses, exceeding the exposures that humans encounter. Hence, transferring results from animal tests to humans may not be readily applicable and can be much more complex to undertake.

For a better illustration of the second stage, we have adapted the quantification method of the WHO (2002). Given the estimated RR, we now estimate the number of cases that are attributable to air pollution (A). To do this, the general empirical formulation can be written as:

$$A_i = b \times \Delta H_j \times Pop_i \times C_i \qquad \text{Eq. 1}$$

Where subscripts, i denotes health endpoints such as mortality, RHA, and RAD and j denotes the pollutant type such as PM_{10}, NO_x, and SO_2. The variable b is the attributable proportion of health effects from air pollution j for the entire population. This can be calculated as, $b = (RR_j - 1) / RR_j$, where in RR is derived from Stage 1 or extracted from other studies. The variable, ΔH_j represents the change in pollutant concentration j, and can be computed in Step 2 of the IPA (dispersion modeling) or readily collected from monitoring stations of each city. The variable Pop represents the population at risk from a given health endpoint i. This variable can be obtained from census data from relevant cities under study. C_i is the population baseline rate of health effect i from exposure to pollutant j. The baseline population rate is the proportion of the exposed population that would experience the health outcome assuming a baseline level (or no effects level) of air pollution. This can be calculated as:

$$C_i = \frac{C_0}{[1 + (RR - 1)(\Delta H_j)]} \qquad \text{Eq. 2}$$

Where C_0 is the observed rate of health effect i, under current exposure and can be obtained from available health statistics. In Stage 2, the resulting output of the Dose Response Assessment is an estimate of the number of cases (Mortality, RAD, RHA) that are attributable to air pollution. Appendix Table 2.1 provides a summary of the DRF studies conducted in Asia.

2.2. Summary of Literature on Dose Response Assessments in Asia

Appendix Table 2.1: Summary of Literature on DRFs

Authors	Year	Location	Study Focus
HEI	2011	India	This study establishes DRFs for Chennai and Delhi by undertaking locally- specific measurements and analyses of data. Results from Chennai (0.4% increase in risk per 10-µg/m^3 increase in PM10 concentration) and Delhi (0.15% increase in risk per 10-µg/m^3 increase in PM_{10} concentration) suggest broadly similar risk of mortality associated with PM_{10} exposure compared with the first four PAPA studies, as well as with multicity studies conducted in the Republic of Korea, Japan, Europe, and North America.
HEI	2010	Multiple	This study enumerates and classifies more than 400 studies identified through a 2007 literature survey. In addition, a systematic and quantitative assessment of 82 time-series studies that estimate the effect of short-term exposure to air pollution on daily mortality and hospital admissions for cardiovascular and respiratory disease. The studies covered in the current review include the coordinated studies of air pollution and daily mortality in four Asian cities conducted as part of HEI's PAPA research program, including qualitative analysis of Asian studies of long-term exposure to air pollution and chronic respiratory disease, lung cancer, and adverse reproductive outcomes.
Wong	2008	Multi-city	This PAPA project examines the effects of short-term exposure to air pollution on daily mortality in Bangkok, Thailand, and in two cities in PRC: Shanghai and Wuhan; and Hong Kong, China. Although the social and environmental conditions may be quite different, it is reasonable to apply estimates derived from previous health effect of air pollution studies in the West to Asia.
Vichit-Vadakan, Vajanapoom and Ostro	2008	Bangkok	This report examines the effects of particulate matter on mortality in Bangkok, Thailand.
Curtis and Rabl	2006	Global	This study analyses the structure of the impact pathway methodology and shows that equations can be simplified for calculating the expectation value of the marginal damage from a point pollution source.
ExternE, European Commission	2005	Multiple	This project focuses on morbidity and mortality risks associated with a range of pollutants. DRFs based on meta-analysis of studies (primarily from the developed world) are also presented.
Thanh and Lefevre	2000	Thailand	This study assesses health impacts of air pollution from power generation using Thailand as a case study.
Chestnut, L.G., Ostro, B.D. and N. Vichit-Vadakan	1997	Bangkok	This study provides a summary of results of health effects and economic valuation studies conducted in Bangkok, Thailand, concerning particulate matter air pollution.
Cropper, et al.	1997	New Delhi	This paper reports the results of a study relating to levels of particulate matter to daily deaths in Delhi, India, spanning from 1991 and 1994. The impacts of air pollution on deaths by age group may be very different in developing countries than in the United States, where peak effects occur among people aged sixty-five and older.
Xu, et al.	1994	Beijing	This paper examines air pollution and daily mortality in Beijing, PRC.

Sources: Multiple studies.

2.3. Evaluation of HEI 2010 Estimates

The HEI Special Report 18, *Outdoor Air Pollution and Health in Developing Countries of Asia: A Comprehensive Review,* is the second full review of the Asian literature on the health effects of air pollution. The first part includes quantitative summarization and comparison of 82 time-series studies that estimate the effect of short-term exposure to air pollution on daily mortality, RHA for cardiovascular and respiratory disease, and RAD (work days or school days), representing nine Asian countries. In addition, two coordinated studies at lower geographic scales are conducted as part of the HEI's PAPA research program– the first one covers air pollution and daily mortality in four Asian cities (Bangkok; Shanghai; Hong Kong, China; and Wuhan) and the other focuses on the two Indian cities (Chennai and Delhi). The second part of the Report covers an in-depth qualitative assessment of Asian studies between long-term exposure to air pollution and chronic respiratory disease, lung cancer, and adverse reproductive outcomes, in which summary effect estimates have not been calculated. It should be noted that we only use the first part of the Report which constitutes systematic summarization and comparison of DRF estimates from 82 individual studies for short-term exposures to air pollution.

We assessed the HEI 2010 estimates on the basis of applicability and reliability considerations (Appendix Table 2.2). First, we assess whether the summary estimates are representative of regional characteristics. This focuses on the quality of studies selected, evidence of publication bias, and study characteristics (e.g., geographic distribution, exposure, health outcome, and study design, among others) (1.1). This also includes how numerical information is extracted from individual studies, consisting of data extraction and recording of numerical information, and lag selection, among other methodological concerns (1.2). Second, we assess whether random-effects summary estimates are robust across all health endpoints in the random effects model. This includes whether empirical results are aligned with a priori expectations. Further, it draws a comparison of findings across other PAPA-SAN studies to check consistency. Also, it draws comparisons to other individual studies to help us interpret severity or magnitude variations from other regions (2.1).

From a careful review of the report following the first criteria, we find that a rigorous screening and selection process of 82 studies has been followed. The HEI analysis builds heavily on its web-based Public Health and Air Pollution in Asia–Science Access on the Net (PAPA-SAN) database and on earlier version of the same report (HEI 2004).[16] The PAPA–SAN database provides a systematic compilation of peer-reviewed scientific studies and literature through 2007, which consists of over 400 Asian studies from three search engines (EMBASE, PubMed, and Web of Science). A list of relevant search strings, which is developed on the basis of health outcomes, pollutants, study site, and study design, is used for screening the studies at various stages of selection. Of more than 400 peer-reviewed articles included in the PAPA–SAN database, 82 studies are retained for the quantitative summarization and comparison of DRF estimates. Variance-weighted summary estimates are calculated when four or more studies have provided estimates for individual pollutant-outcome pairs. Given the sifting process of selection and screening, as well as re-matching of studies for consistency checking (using an external database, University of London's Air Pollution Epidemiology Database or APED), the quality of 82 relevant studies is maintained at all levels. It should be noted that because all 82 studies are peer-reviewed, some degree of publication bias may be present. Results suggest however that there is no statistical evidence of publication bias.[17]

[16] Database can be accessed at http://www.healtheffects.org/Asia/papasan-home.htm/

[17] The meta-analysis may not represent all available evidence as some relevant reports may be published in other formats. This implies that some degree of publication bias may be present, leading to inaccurate standardized effect estimates. Results of the Begg's and the Egger's tests suggest no evidence of publication bias.

Appendix Table 2.2: Qualitative Assessment Matrix for the HEI Estimates.[18]

Criteria	Indicators	Summary/Assessment
1. Applicability 1.1 Quality and Selection of Individual Studies	Are individual studies selected using carefully determined search criteria?	The Current Review of Asian Literature provides summary effect estimates by pollutant concentration from 82 time-series studies spanning from 1980 through 2007. These studies are in essence a meta-analysis, focusing on the daily time series studies of the short-term health effects of outdoor air pollution. The 82 peer-reviewed studies are extracted from HEI's PAPA-SAN and then simultaneously verified using the University of London's APED, each demonstrating a rigorous screening and selection process. The PAPA-SAN database uses a 'general' search strategy, focusing on the identification of air pollution studies on health in Asia, while APED database uses a more specific search strategy, narrowly focusing on the identification of daily time-series studies of the short term health effects of air pollution worldwide. Individual results from PAPA-SAN and APED databases have been simultaneously selected; the latter is used to check for consistency. The screening and selection process of PAPA-SAN and APED can be simplified in the following steps: Step 1: Identifies peer-reviewed studies indexed in three search engines – PubMed, EMBASE, and Web of Science. Search strings are developed on the basis of health outcomes (for both PAPA-SAN and APED), pollutants (for both PAPA-SAN and APED), study site (for PAPA-SAN), and study design (for APED). Step 1 is done simultaneously for both search strategies. Step 2: For PAPA-SAN, selected studies identified in Step 1 are further evaluated using relevant screening criteria such as: (i) site study not relevant to PAPA- SAN, (ii) exposures other than outdoor air pollution, (iii) study only measures concentrations of air pollutants and not health effects, and (iv) study conducted clinical trials with humans or animals. Step 3: Results from PAPA-SAN and APED search strategies are combined. At this stage, those that are time series and provide numerical estimates of the short-term effects of air pollution on health are retained. The usability of these studies depends on the details of the study design, statistical methods, and presentation of results. Hence for inclusion in the final selection, the study must meet all criteria: • The study has at least one year of daily data; • The selected model attempts to control for time and seasonal variation; • The study reports regression coefficients in order to calculate standardized effect estimates; and • The study covers a general population rather than any kind of sub group (e.g., smokers, people with heart disease)

[18] The methodology for deriving the HEI 2010 estimates is discussed in detail in Sections IV and V of the HEI 2010 Report.

Criteria	Indicators	Summary/Assessment
Assessment: There is a structured and systematic screening of literature for both search strategies. All 82 studies are selected from a sifting process of screening and selection using two search strategies, PAPA-SAN and APED. Further noting that since the studies are selected from estimates on human subjects (and not animals, there is some confidence that extrapolation concerns are minimized (See discussion on Stage 2 of Dose Response Assessment, para 9).		
	Are the studies representative of the Asian settings?	Study Characteristics of 82 Studies in terms of geographic distribution, exposure, health outcome, and study design: Geographic distribution: Of 82 studies, majority were conducted in cities in the Republic of Korea (28), followed by the PRC (20), Taipei,China (16), Hong Kong, China (10), Japan (3), Thailand (2), India (1) Singapore (1), and Malaysia (1) Exposure: Majority of the studies estimated the health effects of exposure to both PM and gaseous pollutants Health Outcome: Most common outcome was mortality from all causes of death (38), followed by mortality from cardiovascular disease (24), and all respiratory diseases (24)
Assessment: Very little can be said to fully describe whether 82 studies accurately represent the average effects on health of air pollutants in Asian cities, primarily due to a relative paucity of such studies in the region. Cities in East Asia are overrepresented in this analysis, while those of cities in South and Southeast Asia are sparsely covered. One reason is that an overwhelming majority published to date, still focus on quantitative assessments in more cities in the PRC; Hong Kong, China; Taipei,China; and the Republic of Korea, where data may not be too diffucult to collect.		
	Is there evidence suggesting publication bias?	The meta-analysis, which consists of 82 peer-reviewed studies, may not represent all available evidence as some relevant reports may be published in other formats or may not published at all. Conclusions exclusively based on published studies can sometimes be misleading. This is because positive results have a better chance of being published as studies in peer review journals with higher impact factors. This implies that some degree of publication bias may be present, leading to inaccurate standardized effect estimates. Tests of Begg's and Egger's have been conducted, and results indicate no statistical evidence of publication bias. It should be noted however that evidence of publication bias for respiratory mortality for all ages from PM10 concentration suggests mixed results, with Egger test reporting marginally significant p-value.
Assessment: Collectively, results suggest no evidence of selective underreporting.		
1.2 Estimation Method and Results	How is the numerical information extracted?	The method for abstracting numerical information (data specific to each regression coefficient) appears to be acceptable and does not present itself with any known systematic bias. A data extraction form was completed; each form was divided into two parts: study information and estimate information. Abstracting the latter is critical, as it consists of details about the health outcome and pollutant and all the data necessary for the quantification and standardization (e.g., duration of daily measurement, range used to scale the effect estimates, etc.). The studies have reported numerical information with different functional forms: RRs, regression coefficients, or percent changes in the mean number of events per data as measures of the association between pollutant concentrations and health outcomes. Using MS Access queries, these estimates were standardized in order to make the estimates comparable: percent change in the mean number of daily events associated an increase in pollutant concentration

Criteria	Indicators	Summary/Assessment
Assessment:: Data extraction is relatively systematic. However, the Report is silent as to how results were extracted from studies, e.g., whether manually entered and recorded in the Access Database.		
	How are lag structures selected (e.g., single, cumulative, etc.) and multiple studies treated?	The authors use a simplified approach in selecting time lags and numerical information to use (e.g., single city with multiple studies). Selecting the lag times for each study is another important component in the analysis of health effects and air pollution, primarily due to the fact that lag times explicitly represent the overall effect size of pollutant exposure in health outcomes (e.g., single, cumulative, distributive). The Report adopts simplified criteria for selecting the lag time, if more than one lag is presented: the lag time with either the highest statistical significance or largest effect magnitude, irrespective of direction of the effect estimate. For a single city with multiple studies, the authors have used the latest available numerical information on the basis that it would most likely reflect current analytic techniques and recent pollution concentrations.
Assessment: The selection of time lags and the numerical information to include (e.g., in the case of multiple studies of a single city), there is no generally accepted standard for this process and the study appears to have set out a fair justification for doing so.		
2. Reliability 2.1 Method of summarizations	Does the model consider uncertainty?	Both fixed-effects and random-effects models are employed for the pollutant-outcome pairs. The authors have tested for evidence of differences in result size or direction, and hence, a random-effects model is utilized to incorporate this uncertainty. Potential reasons for variations include composition, physical characteristics, and source of pollution to which the population is exposed, variation in meteorologic conditions. After accounting for uncertainty, the random-effects model yielded favorable results i.e., coefficient estimates that were substantially different from those derived from the fixed-effects model. The WP currently utilizes summary estimates from the random-effects model.
	Are summary estimates robust?	Results are moderately robust with respect to a priori expectations. Notably, all pollutants were positively associated with adverse health effects (although there are very few studies that have estimates and/or lower confidence limits that are below zero). Is the magnitude of effects consistent with other studies? It is not necessarily the case. One possible explanation is the variation in exposure measurement errors (e.g., when measurement errors are less (more) differential with respect to population at risk, the estimate is likely to be biased downward (upward).
	Are the estimates comparable to other studies?	PAPA Report from Delhi (Rajarathnam et al 2010): Result falls below the reported estimate from the current review. For a 10µg/m3 increase in PM10 concentration, the daily mean for all-natural-cause mortality in all age groups increased by 0.15%. This is substantially lower than what is reported in the current review (0.23%).

Criteria	Indicators	Summary/Assessment
		PAPA Report from Chennai (Balakrishnan et al. 2010): Result falls above the reported estimate from the current review. For a 10µg/m3 increase in PM10 concentration, the daily mean for all-natural-cause mortality in all age groups increased by 0.40%. This is substantially higher than what is reported in the current review (0.23%). PAPA Report in four Asian cities (Bangkok; Shanghai; Hong Kong, China; and Wuhan): All summary estimates from the PAPA study for all-natural-cause and cardiovascular mortality are higher than those calculated from current review.
Assessment on reliability: Broadly, summary estimates are consistent with expectations in terms of signs though there are a few exceptions (i.e., selected individual studies in morbidity). Compared to other regions, the estimates fall somewhat in between the estimates presented in other studies, hence generally consistent to expected magnitude of effect (estimate of RR).		

Source: *Summarized versions of HEI 2010 Report Methodology Section.*

With respect to the geographic distribution of the studies, we find that cities in East Asia were overrepresented in this analysis, while those of cities in South and Southeast Asia are sparsely covered. Of 82 time-series studies on short-term exposure, 28 studies are conducted in the Republic of Korea, followed by the People's Republic of China (PRC) (20); Taipei,China (16); Hong Kong, China (10); Japan (3); Thailand (2); India (1); Singapore (1); and Malaysia (1). One obvious reason is that an overwhelming majority published to date, focusing on quantitative assessments between exposure to outdoor air pollutants and health effects, covered more cities in the PRC; Hong Kong, China; Taipei,China; and the Republic of Korea. Although the existing studies do not yet represent the full range of Asian settings, empirical evidence suggest that when estimates from individual studies are combined into summary estimates they resemble results from more extensive, coordinated multicity studies conducted in Europe and North America (HEI 2004). We note that while results in other Asian regions might be broadly similar to those in East Asia, the levels and composition of air pollution, source types, and factors related to population health and socioeconomic development may result in differences in the health effects of short-term exposure to outdoor air pollution. This can be circumvented in Stage 2 of Dose Response Assessment, where in local variables such as background rates, numerical data on population, health data.

In terms of reliability considerations, mortality and morbidity outcomes are reported in separate contexts. It should be noted that mortality is more straight-forward in that it focuses on one endpoint (e.g., death). Morbidity impacts on one hand, manifest through various health endpoints, such as chronic bronchitis, chronic cardiovascular disease, RHA, cardiovascular hospital admission, emergency room visits (ERVs), consultations with physician for asthma episode, RAD, and WLD, among others. For this reason and that of a simple limitation of research possibilities, we only report DRF estimates for selected morbidity endpoints.

The empirical results for mortality are moderately robust after accounting for heterogeneity and adjusting for uncertainty for both mortality and morbidity outcomes. Notably, most of the associations are positive and represented an increase in the mean daily mortality. As shown in Table 2 of the main text, the daily mean for all-natural-cause mortality in all age groups increased by 0.27%; respiratory mortality, by 0.86%; and cardiovascular, by 0.36% for every 10μg/m^3 increase in PM_{10} concentration (Rajarathnam et al. 2011). Comparing these magnitudes to other PAPA-SAN studies, we find that results, in particular PM and NO concentrations on all-natural-cause mortality fall somewhat in between these studies. The PAPA reports for two Indian cities suggest DRF values for PM concentrations of 0.15% for Delhi and 0.40% for Chennai respectively; and NO concentration of 0.84% for Delhi (Rajarathnam et al. 2011; Balakrishnan et al. 2011). Relative to estimates from the coordinated PAPA report in four Asian cities (Bangkok; Shanghai; Hong Kong, China; and Wuhan), all HEI summary estimates for all-natural-cause and cardiovascular mortality are smaller. Further, in the study conducted by Vichit-Vadakan et al. (2008) for PM concentration in Bangkok, their results suggest higher estimates for overall non-accident, cardiac and respiratory mortality are 1.3%, 1% and 1.9% respectively.

Further drawing comparisons to other regions, the same observation tends to apply— in that, the HEI estimates fall within the range of estimates reported in these studies. For instance, in the study of Anderson et al. (1997) for selected European cities, they have a reported indicate a value of 0.6% for PM_{10}. Likewise in a study conducted by Daniels et al. (2000) in 90 US cities, they have reported a value of 0.5% for PM_{10}. This observation is broadly consistent with another study comparing Asian and Western cities. In a study conducted by Wong (2008), it is concluded that the health effects of PM_{10} and gaseous matter (SO_2 and NO_2) in Asia are similar or higher than those in North American and Western European cities. The study computes the increased risk rate of all-cause (natural deaths) mortality, cardiovascular mortality and respiratory mortality due to PM_{10}. The values calculated are 0.55%, 0.58% and 0.62% for PM_{10}.[19]

In a study conducted in Hong Kong, China, Wong et al. (2006) have reported significant associations between first visits for URTI (upper respiratory tract infections) and increased concentrations of NO_2, O_3, and PM. They have reported that excess risk is highest for NO_2 (3.0%), followed by O_3 (2.5%), $PM_{2.5}$ (2.1%), and PM_{10} (2.0%). In a study conducted by Patankar and Trivedi (2011) in Mumbai, they have reported an increase in risk of 0.06% and 0.2% for respiratory symptoms associated with increases in PM_{10} and NO_2. These studies and others indicate that a broad range of values are provided in different studies and it is important to carry out a detailed analysis of specific conditions in each of the studies, before arriving at a summarized estimate of dose response. Overall, we find that the individual estimates for morbidity endpoints conform to expectations but there are few exceptions. For example, estimates from some individual studies reported lower confidence limits that are below zero (e.g., cardiovascular admissions related to NO_2.

We report a range of estimates from individual studies, in cases where summary estimates are not calculated for morbidity endpoints. Only summary estimates of RHA with respect to NO and SO are reported, as well as their 95% confidence intervals. The rest of the endpoints report a range of estimates, in which the lower and upper bounds have been extracted from individual studies included in the HEI list. For instance, the reported range of estimates of RHA (all respiratory causes) for PM concentration uses individual estimates reported from three studies, Chang et al. (2005) in Taipei,China, 0.1%; Wong et al. (1999) in Hong Kong, China, 0.6%; and Leem et al. (1998) in the Republic of Korea, 1.8% (Appendix Table 3). Taking stock from these three studies that quantitatively assess RHA for all respiratory causes, we have identified a range of 0.1% to 1.8%.

We have identified the individual studies through the forest plots that are presented in the HEI 2010 Report.[20] For all morbidity endpoints for which no summary estimate is available, a total of 10 individual studies have reported estimates. We have attempted to collect all 10 studies from epidemiological web engines, in order to check the exact estimates rather than simply conduct an eyeball estimation on the forest plots. However, we are only able to obtain four studies due to data access limitations. In cases where we do not have the actual studies, an eyeball estimation on the forest plots has been conducted.[21]

[19] With respect to morbidity outcomes, several studies (Peters et.al 1999, Lin et al 2002, Ostro et.al 1993) present the increased incidence of morbidity in terms of odds ratio, which can be approximated to the relative risk at small values of probability, but not otherwise.

[20] Forest plots are diagrams containing all effect estimates for pollutant-outcome pairs from all relevant time series studies. Each relevant study is represented by a horizontal number line. The estimates are normally presented as percent change in the mean number of daily deaths (for mortality) and hospital admissions (for morbidity) including confidence intervals.

[21] Because we are not able to access all studies contained in the forest plots, the mean percent changes and their confidence intervals are visually estimated following interval values at the x-axis.

APPENDIX 3: ECONOMIC VALUATION

This appendix supports the main text discussion of the Step 4 of the IPA, where in we describe the approach used for estimating the monetary values of health impacts. Appendix 3 is divided into four sections: Section 3.1 describes the concept of Value of Statistical Life (VSL) and a summary of Asian studies that have conducted VSL as well as benefit transfer. Section 3.2 discusses the literature on valuation of morbidity.

3.1 Value of Statistical Life (VSL)

Value of statistical life represents the willingness to pay (WTP) of an individual to reduce the risk of mortality by a specified amount. This definition indicates that the willingness to pay depends on the magnitude of risk reduction involved and the risk related perceptions of the individual. Societal valuations of WTP can also represent the value placed by the society on reducing the risk of death. For many years, economists have studied individuals' preferences over mortality and morbidity risk and tried to infer preferences in surrogate markets such as property market and labor market. These investigations resulted in hedonic wage and hedonic pricing models.

In practice, the WTP for life risk reduction can be estimated using stated preference methods or revealed preference methods. Stated preference method requires a WTP survey to be conducted, to directly elicit the WTP for life risk reduction. Given various difficulties involved in eliciting preferences through surveys, economists prefer revealed preference methods (Gunatilake 2003). Surrogate market data, (i.e, prices) are used to infer WTP in revealed preference methods. In the case of mortality valuation, hedonic wage model is used to infer the WTP for life risk reduction. The hedonic wage approach treats job as bundles of characteristics such as working conditions and levels of risks of accidental injury and death. Employees are described by the amount they require as compensation for different risk levels while employers are characterized by the amount they are willing to offer to the workers to accept different risk levels. An acceptable match occurs when the preferred choice of an employee and an employer matches each other. Thus, the actual wage embodies a series of hedonic prices for various job attributes including accidental risk and other prices for worker characteristics (Arnold and Nichols 1983, Viscusi 1993; Viscusi and Joshep 2003).

Suppose that there are m indicators of a worker's personal and job attributes other than risk levels (p) and risk denoted by vector $c = (c_1, c_2, \ldots c_n)$. Let w represents the annual earnings, then w(p,c) reflects the market equalizing wage function. Controlling for other aspects of the job would provide an estimate of wage premium that workers are willing to accept for the given risk. In estimating the VSL, a hedonic wage equation incorporating worker's personal characteristics, job characteristics, and probability of work related fetal and non-fetal risks is estimated. The VSL can be estimated using the coefficient of fetal risk variable of the regression results (see Shanmugam and Madheswaran 2011 for details).

Estimation of a hedonic wage equation is a time consuming and data intensive process. Given the availability of time and resources are constrained for evaluating the mortality costs associated with a project, 'benefit transfer' method if often used. Summary of the literature on application of VSL is given in Appendix Table 3.1.

Appendix Table 3.1: Summary of Literature for VSL

Authors	Study Location	Study Focus	Valuation Technique	Results
Rozan (2004)	France and Germany	This study is commissioned in order to test the reliability of Value Transfer method under intratemporal and intrasite conditions. A CV was simultaneously carried out under similar conditions on two neighboring sites: Strasbourg (France) and Kehl (Germany). The underlying principle of this test is to obtain a transferred mean WTP for the policy site that is not significantly different from the mean WTP for the policy site obtained directly.	Contingent valuation (CV)	• For the VT exercise from Strasbourg to Kehl, the two mean WTP are significantly different (the transferred WTP falls outside the 95% Confidence Interval for the direct WTP), in which the transferred WTP was significantly smaller than the direct one. Estimated error rate is about 30%. • For the VT exercise from Kehl to Strasbourg, the transferred WTP falls outside the 95% Confidence Interval for the direct WTP, in which the transferred WTP was significantly higher than the direct one. Error rates are 16% to 30% for smokers and nonsmokers. • In summary, results from VT imply high error rates. As the survey results suggest, Kehl residents indicate a much higher price for their state of health and air quality than those of Strasbourg residents. One reason cited is the stronger sensitivity to environmental problems in Germany. • The authors, in general, do not advocate the use of VT, however this may depend on purpose. If the transferred estimate will be used for CBA and policy decisions, then it may be warranted provided that an error rate is 15% or less. However if the purpose is for establishing a reference amount for estimating compensation, then application of VT may not be acceptable.
Viscusi and Aldy (2003)	Global	This is a meta-analysis report based on more than 60 studies of mortality risk premiums and 40 studies with estimates for injury risk premiums. The report examines econometric issues in hedonic labor market literature, and the effects of age on the VSL. This study examines the effects of age and the role of income differences in generating the variation in VSL estimates. The study attempts further to answer the question 'what other factors	Hedonic wage (HW)	• The VSL is in the range of $4 million to $9 million, based on the estimated using US labor market data. These are in line with those values generated by US product market and housing market studies. • Income elasticity should be positive on theoretical grounds. However, empirical estimate of this elasticity is needed in order to extrapolate the VSL estimates across different contexts. Results of the meta-analyses of VSL

Authors	Study Location	Study Focus	Valuation Technique	Results
		may influence the transfer of mortality risk valuation estimates from journal articles to policy evaluation in different contexts?		estimates suggest a range of income elasticity from 0.50 to 0.60. • Results also suggest that labor market union members have greater risk premiums than non-members. • In terms of age, they find the effects of age consistent with a priori expectations– VSL decreases with age. • In all, this implies that heterogeneity in VSL estimates based on union status and age indicate that the VSL not only varies by income but also by labor market dimensions. Hence, the existence of such heterogeneity provides a cautionary note for policy.
Dekker et al. (2011)	Global	This is a meta-analysis of 26 Stated Preference Studies (using CV) that empirically estimates correction factors for 'out of context' VT contexts. This builds on the premise that estimates need to be properly corrected when transfers are made between mortality risk contexts (e.g., transferring VSL estimate from road safety to air pollution context).	CV	• There is considerable variation in VSL estimates both within and between risk contexts (i.e., air pollution, road safety, and general context) with risk perception affecting WTP and the size of risk reduction influencing the VSL estimate. The VSL ranges within air pollution, road safety, and general context between \$0.13 and \$5.43 million, \$0.73 to \$33.58 million, and \$0.55 to \$8.91 million (in 2004 PPP converted prices). • The study proposes a correction factor of 1.8 when transferring VSL estimates from road safety to air pollution.
Alberini et al. (1997)	Taipei,China	This study estimates the WTP to avoid the recurrence of a respiratory episode from air pollution using data from Taipei,China. It also examines the VT method between two different economies (Taipei,China and the US). This was done by first using an adjusted unit value transfer and then the value function transfer. The WTP to be transferred is from previous studies in the US (i.e., Loehman et al. and Tolley et.al.). These transferred estimates were compared with the direct WTP estimate. For an adjusted unit value transfer, income elasticities of 1.0 (simplest	CV	• Income elasticity of WTP is estimated to be about 0.41. • WTP to avoid illness increased with duration of illness, with the number of symptoms experienced, and with education and income. Results suggest that WTP does not vary systematically with age and insurance status. • Estimates of WTP also imply that WTP tends to be lower in a low income country than in a high income country, however less than proportionally to the income differential. • Results of the VT exercise using the adjusted unit value transfer

Authors	Study Location	Study Focus	Valuation Technique	Results
		assumption) and 0.41 (estimated from the study itself). For the value function transfer, the WTP function estimated for the study was used to predict the WTP in the United States.		technique (adjusting for income differentials) and value function, suggest that none of these techniques yield unambiguously superior results. The authors suggest that more credible transferred WTP estimates may be obtained by designing original valuation studies that are of high quality, which in turn can support the future use in a VT analysis.
Hammitt and Zhou (2006)	PRC	This is a CV study on the estimation of WTP of three health risks from three regions in PRC. These health risks constitute two morbidity effects (cold and chronic bronchitis) and mortality.	CV	• Estimates for the mean VSL range from $15,000 -$30,000 in Anquing, $45,000-$60,000 in Beijing, and $100,000-$180,000 in rural areas. These estimates are sensitive to modeling choices and location. The estimates for PRC are between 100 to 1,000 times smaller than the US estimates (one possible reason cited is that the mortality-risk reduction presented in the CV questions (1 or 2 per 1,000) is much larger than the risk reduction typically presented in CV studies (parts per 10,000). Hence, the estimates of VSL for PRC should at best be treated as a lower bound. • As results suggest, WTP tends to be higher for younger, more educated, and higher-income earning respondents, as what one would expect. Interestingly however, the variable risk reduction was not significantly different from zero, suggesting that the magnitude of the risk reduction may not be an important factor for respondents. Income elasticity ranged from 0.06 to 0.20 for the three regions, the lowest recorded in rural areas.
Vassanadumrongdee and Matsuoka (2005)	Thailand	This study presents two CV surveys in Bangkok, estimating the individuals' WTP to reduce mortality risk associated from two risk contexts: air pollution and traffic accidents.	CV	• Results suggest that WTP to reduce mortality risk arising from air pollution is a function of several factors: degrees of dread, severity, controllability and personal exposure. Whereas WTP to reduce traffic accident risk is driven by perceived immediate occurrence. Hence, this suggests that respondents view these risks differently.

Authors	Study Location	Study Focus	Valuation Technique	Results
				• Estimates of VSL are nonetheless, comparable between the two risks. Estimate of VSL from air pollution ranges from $0.74 million to $1.32 million, while those of traffic accidents fall between $0.87 million to $1.48 million. This suggests that the role of risk perception has little (to no) significant impact on the VSL, which is consistent with previous empirical findings.
Madheswa-ran (2007)	India	This study estimates the VSL using hedonic price model. The goal of this study is to estimate VLS that reflects Indian risk preferences, based on a survey of 550 workers in Chennai and 535 workers in Mumbai.	Hedonic Price (HP)	• Estimate of VSL is about Rs.15 million ($340,000). The Value of statistical injury ranges from Rs. 6,000 to Rs. 9,000. These estimates can be used reference points for assessing compensation.

Source: Various studies.

3.2. Cost of Illness and Willingness to Pay

Willingness to pay (WTP) for reducing the risk of morbidity is much larger than the cost of illness (COI). Appendix Table 3.2 summarizes the literature on the relationship between WTP and COI.

Appendix Table 3.2: Summary of Literature Comparing COI and WTP to Avoid Health Illness associated with Air Pollution

Authors	Study Location	Study Years	Risk context	Study Focus	Results	WTP/COI ratio
Alberini and Krupnick (2000)	Taipei,China	1991–92	Minor health damages associated with air pollution (PM)	An empirical exercise comparing COI and WTP estimates, based on a combined physical and monetary valuation study (epidemiological & economic). The epidemiological component is a cohort study of survey participants who were asked to fill out daily questionnaires about 19 minor respiratory-related symptoms (e.g., chest discomfort, coughing, wheezing, sore throat, cold, flu, and others) and activities undertaken to relieve these symptoms. The economic component is a contingent valuation (CV) survey where in respondents report information about their WTP to avoid an episode of illness similar to the one they had most recently experienced (this is a departure from other WTP studies in which the commodity to be valued is defined by the respondent, rather than by the researcher). COI and WTP estimates are compared in order to first validate economic theory that WTP should be greater than COI. Then, the study calculates a scaling factor–the fraction of COI constituting total WTP. Finally, the study compares this scaling factor derived from a developing economy data (Taipei,China) with that of the WTP/COI ratios estimated for the United States and whether there are significant deviations present.	• Total COI is broken down into direct expenditures (i.e., doctor cost and prescription medication expenses) and indirect expenses (i.e., lost of earnings). Depending on the level of pollution, COI ranges from \$536,689 (1992 \$) at very low levels of PM (25μg/m^3) to as high as \$1, 048,775.0 at 350μg/m^3. These estimates do not include the value of travel time to, or waiting time at, the doctor's office. • WTP estimates range from US \$794,733 at very low levels of PM to as high as \$1,048,775. WTP increases with income and education, and is typically higher for individuals who have suffered from serious respiratory illnesses, or chronic illnesses, suggesting increasing marginal disutility of illness. • Combining the COI and WTP estimates, the WTP/COI ratios range from 1.48 at very low levels of PM to 2.26 at the highest PM readings (350μg/m^3). • Overall, the empirical results suggest that WTP estimates exceed COI, which is what one would expect. Second, the WTP/COI ratios for Taipei,China are in line with those computed for the United States, despite differences economic, institutional, and cultural differences.	• 1.88 (overall average computed ratio for all 14 estimates by pollutant level) • 14 WTP/COI ratios depending on pollutant concentration: 1.48 to 2.26 • Average computed ratios by pollutant levels: a. 25μg/m^3 to 100μg/m^3 –1.57 b. 101μg/m^3 to 150μg/m^3 –1.74 c. 151μg/m^3 to 300μg/m^3–2.00 d. Above 300μg/m^3–2.25

Authors	Study Location	Study Years	Risk context	Study Focus	Results	WTP/COI ratio
				Once computed, the scaling factor may be used to infer the total WTP by first obtaining data of policy area for the computation of COI, and then by multiplying the COI by this scaling factor.		
Dickie and Gerking (1991)	United States (Los Angeles, California)	1985-86	Air pollution (ozone concentration) on chronic obstructive respiratory disease and compromised respiratory function	Using random effects probit models, estimates of doctor visits are obtained as a function of pollution, proxies for the individual's stock of health, and price of medical attention, and these were used to compare WTP (area under the curve between the specified high and low ozone levels) with the cost of doctor visits. Maximum daily one-hour ambient concentrations (CO, NO_2, O_3, and SO_2) are used, as provided for by epidemiological evidence that acute health problems likely that would likely to induce a visit to the doctor may be more closely related to peak than to average concentrations.	• WTP and the medical expense estimates are nonlinear functions of the upper and lower bound ozone levels. WTP estimates can be twice or four times as large as the expenditure on doctor visits, depending on the probit model specification upon which the demand curve is based. • For a probit estimation with restriction that there is no variance of the individual-specific error component, the average WTP/COI ratios range from 1.94 at less strict ozone control (maximum peak at 12pphm) to 2.20 with stricter ozone control (maximum peak at 9pphm). • For a probit estimation that allows for an interaction variable–maximum daily ozone concentrations and whether the respondent reported physician diagnosed chronic lung diseases (i.e., Asthma, chronic bronchitis, emphysema, or lung cancer), the average WTP/COI ratios range from 3.76 at less stricter ozone control (maximum peak at 12pphm) to 4.09 with stricter ozone control (maximum peak at 9pphm). • Although the interaction variable is not statistically significant, the WTP estimates are higher for this estimation. This somewhat implies that medical care demand increases with the joint effect between ozone pollution and presence of physician diagnosed chronic lung disease.	• 3.00 (overall average computed ratio for all 8 estimates by city, ozone concentration, and estimation model) • 8 WTP/COI ratios depending on city, ozone concentration, and estimation model: 1.98 (ozone concentration of 12pphm; no interaction variable; Burbank) to 4.17 (ozone concentration of 9pphm; with interaction variable; Burbank) • Average range between Burbank and Glendora: 1.94 (ozone concentration of 12pphm; no interaction variable) to 4.09 (ozone concentration of 9pphm; no interaction variable), depending on the probit model specification upon which the demand curve is based

Authors	Study Location	Study Years	Risk context	Study Focus	Results	WTP/COI ratio
Rowe and Chestnut (1985)	United States (Los Angeles, California)	1983	Air pollution (carbon monoxide) on heart patients with asthma	The study has two components: epidemiological and economic. For the epidemiological component, participants are asked to maintain a daily record of his or her asthma symptoms over an eleven month period. For the economic component, this constitutes a survey for which the purpose is to identify ways in which asthma affects people's well-being and to estimate economic measures of changes in well-being associated with changes in the frequency of asthma symptoms. The primary goal of the study is to compare COI estimates with WTP estimates. To estimate medical expenditures, data on medical supplies, equipment and special treatment programs were collected, as well as information on doctor and hospital visits were obtained.	• Respondents were asked to rank perceived benefits which they may gain from reduced asthma, discomfort and asthma effects on leisure and recreation activities were ranked first, following medical costs and work loss (cumulatively known as COI). This implies that if changes in disutility arising from discomfort and leisure activity effects are valued more highly than changes in medical costs and work loss, then cost of disutility is therefore higher and cannot be simply ignored, further reflecting that WTP exceeds COI. • When comparing these two estimates, one limitation is that the WTP estimates are based upon a 50% change in bad asthma days while the estimated reduction in medical costs are based upon a 50 percent change in severity measured as the sum of monthly frequency times the intensity of asthma symptoms. • The ratio obtained for WTP/COI is 1.61 from an individual perspective. This is expected to understate the true ratio due to differences in the manner in which the COI and WTP values were estimated. In an earlier study (Rowe & Chestnut 1984), this was estimated at 3.7 or about roughly one-third of total WTP for changes in asthma severity. • The authors have also computed the WTP/COI ratio from a social perspective, by adjusting the individual WTP and COI values using estimated social costs and benefits, but such difference (between social and individual) can only be accounted for medical cost and not for perceived work loss costs. In	• WTP/COI ratio of 1.61 (individual perspective) • WTP/COI ratio ranges from 1.31 to 2.35, taking into account social perspective • WTP/COI recommended ratio is in the range of **1.5 to 3.00**.

Authors	Study Location	Study Years	Risk context	Study Focus	Results	WTP/COI ratio
					summary, the social COI is estimated at about double the individual's COI. The social WTP is also expected to exceed the individual WTP. Following assumption on social incurred medical costs and work loss and that WTP by others in society to reduce an individual's asthma is zero, the WTP/COI ratio ranges from 1.31 to 2.35. When society's WTP equals half of the individual' WTP, the social WTP/COI ratio increases to a range 1.55 and 2.6. • The authors recommend using WTP/COI ratio of 1.5 to 3.00 for asthma severity. This ratio is not representative of other types of illnesses. • For minor health effects such as eye and throat irritation, WTP/COI ratio may be higher than what is reported. While smaller for major illnesses such as angina or cancer.	
Chestnut, et al. (1996)	United States (Irvine, California)	1985	Air pollution (carbon monoxide) on heart patients with angina	This study examines both the estimated COI and WTP. To estimate WTP, the study used two approaches: one is via avertive behavior measurement and the other, by CV method. All estimates are compared to examine whether significant variations are present.	• Average COI for the sample is $14,359, constituting total medical costs and income lost. Changes in COI are negligible in small changes in angina frequency. • Estimates of WTP to avoid angina from CV questions were of similar general magnitude as estimates calculated from patient reports of actual expenditures and the perceived episodes avoided. • Mean estimated averting expenditure per angina episode avoided was $38, whereas mean direct WTP response was $28 per additional episode avoided.	• No WTP/COI ratio computed

Appendix Table 3.3: Computation of Average Scaling Factor (WTP/COI ratio)

Illness (Degree of Severity)	Estimate	Range	Reference	Average (lower bound)	Average (upper bound)	Average Scaling Factor
Minor Health Problems	1.88	1.48 to 2.26	Alberini and Krupnick (2000)			
Asthma and other compromised respiratory function	3.00	1.94 to 4.09	Dickie and Gerking (1991)			
	1.61	1.50 to 3.00	Rowe and Chestnut (1985)	1.64	3.12	2.16
Angina or lung cancer	–	–	Chestnut et al. (1996)	–	–	–

Note: Dash (–) denotes that no value has been provided.

APPENDIX 4: CASE ILLUSTRATION

This appendix presents background details and data used for the case illustration. Section 4.1 presents the background data used for Gaussian-plume dispersion model. Section 4.2 presents background information on mortality and morbidity.

4.1. Gaussian-plume dispersion parameters

Lateral Dispersion (σ_y)

$\sigma_y = (x * \alpha)/\sqrt{}(1+0.0001*x)$

Where

very unstable conditions $\alpha = 0.22$
moderately unstable conditions $\alpha = 0.16$
slightly unstable conditions $\alpha = 0.11$
neutral conditions $\alpha = 0.08$
somewhat stable conditions $\alpha = 0.04$
stable conditions $\alpha = 0.04$

Vertical Dispersion (σ_z)

very unstable conditions $\sigma_z = x*0.20$
moderately unstable conditions $\sigma_z = x*0.12$
slightly unstable conditions $\sigma_z = x*0.08 / \sqrt{}(1+0.0002*x)$
neutral conditions $\sigma_z = x*0.06/ \sqrt{}(1+0.00015*x)$
somewhat stable conditions $\sigma_z = x*0.03 / (1+0.0003*x)$
stable conditions $\sigma_z = x*0.016 / (1+0.0003*x)$

Effective Height (H_{eff})

$H_{eff} = H + (1.6*e^{(\ln f_o)/3} * e^{(2*\ln(3.5*x_o)/3)}) / u$

Where

H = physical height of the stack (in metres)
u = wind speed (in m/sec)
f_o and x_o are given by the formula
$f_o = 3.12*0.785*v_o*d2*(t_o-t_1)/t_o$

and

if $f_o > 55$, then $x_o = 34* e^{(0.4*\ln f_o)}$
if $f_o <= 55$, then $x_o = 14* e^{(0.625*\ln f_o)}$

Where

vo = gas exit velocity (in m/sec)
d= stack diameter (in metres)
to = gas exit temperature (in degree K)
t1 = ambient exit temperature (in degree K)

4.2. Background Information on Mortality and Morbidity

Appendix Table 4.1: Background Mortality Rates in India, Cause and Age-wise Split, 2001-03

Endpoint	Mortality Rate (per 1000)
All cause (non-accident)	6.66
Cardiovascular	1.406
Respiratory/COPD	1.11
All cause (>65)	0.74
All cause (non-cause and age specific)	3.404

Source: National Communication (2011).

Appendix Table 4.2: Baseline Mortality by City, Base Results

		Baseline deaths				
City	Population	Cardiovascular	Respiratory	All cause, >65	All cause exc. Cause/ age specific	Total
Chennai	6,540,462	9,196	7,260	4,840	22,264	43,559
Pondicherry	505,959	711	562	374	1,722	3,370
Kancheepuram	188,733	265	209	140	642	1,257
Cuddalore	158,634	223	176	117	540	1,057
Thiruvannamalai	130,567	184	145	97	444	870
Arakonam	78,686	111	87	58	268	524
Thindivanam	67,737	95	75	50	231	451
Chengalpattu	62,852	88	70	47	214	419
Arani	60,815	86	68	45	207	405
Thiruvallur	45,732	64	51	34	156	305
Melvisharam	36,757	52	41	27	125	245
Sriperumbudur	16,156	23	18	12	55	108
Ananthapuram	6,138	9	7	5	21	41
Vandavasi	29,620	42	33	22	101	197
Total	7,928,848	11,148	8,801	5,867	26,990	52,806
%		*21%*	*17%*	*11%*	*51%*	–

Source: Authors' estimates.

Appendix Table 4.3: Cases Occurred per Person (Mortality), Base Results

City	SO_2	NO_2	PM_{10}
Chennai	0.0004	0.0021	0.0005
Pondicherry	0.0008	0.0041	0.0010
Kancheepuram	0.0006	0.0031	0.0007
Cuddalore	0.0005	0.0025	0.0006
Thiruvannamalai	0.0003	0.0016	0.0004
Arakonam	0.0004	0.0021	0.0005
Thindivanam	0.0011	0.0056	0.0013
Chengalpattu	0.0011	0.0056	0.0013
Arani	0.0004	0.0021	0.0005
Thiruvallur	0.0004	0.0021	0.0005
Melvisharam	0.0003	0.0016	0.0004
Sriperumbudur	0.0006	0.0031	0.0007
Ananthapuram	0.0005	0.0025	0.0006
Vandavasi	0.0008	0.0041	0.0010

Source: Authors' estimates.

Note: Cases occurred per person is computed by multiplying mean DRF to the incremental pollution concentration.

Appendix Table 4.4: Cases Avoided/Occurred per Person (Morbidity), Base Results

City	SO_2, RHA	NO_2, RHA	PM_{10}, RHA	PM_{10}, WLD
Chennai	0.0003	0.0019	0.0004	0.0009
Pondicherry	0.0006	0.0038	0.0008	0.0018
Kancheepuram	0.0004	0.0029	0.0006	0.0014
Cuddalore	0.0004	0.0023	0.0005	0.0011
Thiruvannamalai	0.0002	0.0015	0.0003	0.0007
Arakonam	0.0003	0.0019	0.0004	0.0009
Thindivanam	0.0008	0.0053	0.0010	0.0025
Chengalpattu	0.0008	0.0053	0.0010	0.0025
Arani	0.0003	0.0019	0.0004	0.0009
Thiruvallur	0.0003	0.0019	0.0004	0.0009
Melvisharam	0.0002	0.0015	0.0003	0.0007
Sriperumbudur	0.0004	0.0029	0.0006	0.0014
Ananthapuram	0.0004	0.0023	0.0005	0.0011
Vandavasi	0.0006	0.0038	0.0008	0.0018

Source: Authors' estimates.

APPENDIX 5: HEALTH COST ESTIMATES

Appendix Table 5.1: Summary of Literature on Estimated Cost of Pollution

Study	Country	Fuel	External Cost Range (US cents/kWh)	Method
Schuman and Cavanagh (1982)	US	Coal	0.06-44.07	Abatement cost
Hohmeyer (1988)	Germany	Fossil fuels	2.37-6.53	Damage cost (top- down)
Chernick and Caverhill (1989)	US	Coal	4.37-7.74	Abatement cost
		Oil	4.87-7.86	
		Gas	1.75-2.62	
Bernow and Marron (1990; Bernow et al. (1991)	US	Coal	5.57-12.45	Abatement cost
		Oil	4.40-12.89	
		Gas	2.10-7.98	
Friedrich and Kallenbach (1991); Friedrich and Voss (1993)	Germany	Coal	0.36-0.86	Damage cost (bottom-up)
Ottinger et al. (1997)	US	Coal	3.62-8.86	Damage cost (bottom-up)
		Oil	3.87-10.36	
		Gas	1.00-1.62	
Putta (1991)	US	Coal	1.75	Abatement cost
Hohmeyer (1992)	Germany	Fossil fuels	11.12	Damage cost (top-down)
Pearce et al. (1992)	UK	Coal	2.67-14.43	Damage cost (top-down)
Cifuentes and Lave (1993); Parformak (1997)	US	Coal	2.17-20.67	Abatement cost
Oak Ridge National Laboratory and Resources for the Future (1994-98)	US	Coal	0.11-0.48	Damage cost (bottom-up)
		Oil	0.04-0.32	
		Gas	0.01-0.03	
Regional Economic Research Inc. (1994)	US	Oil	0.03-5.81	Damage cost (bottom-up)
		Gas	0.003-0.48	
European Commission (1995)	Germany	Coal	2.39	Damage cost (bottom-up)
		Oil	3.00	
European Commission (1995)	UK	Coal	0.98	Damage cost (top-down)
		Gas	0.10	
Pearce (1995)	UK	Coal	3.02	Damage cost (top-down)
		Gas	0.49	
Rowe et al. (1995)	US	Coal	0.13	Damage cost (bottom-up)
		Oil	0.73	
		Gas	0.22	
Van Horen (1996)	South Africa	Coal	0.90-5.01	Damage cost (bottom-up)
Bhattacharyya (1997)	India	Coal	1.36	Damage cost (bottom-up)
Ott (1997)	Switzerland	Oil	12.97-20.57	Damage cost (top-down)
		Gas	8.85-13.22	
Faiij et al (1998)	The Netherlands	Coal	3.98	Damage cost (top-down)
Faiij et al (1998)	The Netherlands	Coal	3.84	Damage cost (bottom-up)
European Commission (1999)	Austria	Gas	0.88	Damage cost (bottom-up)
European Commission (1999)	Belgium	Coal	3.22-67.72	Damage cost (bottom-up)
European Commission (1999)	Denmark	Gas	0.99-11.19	Damage cost (bottom-up)

Study	Country	Fuel	External Cost Range (US cents/kWh)	Method
European Commission (1999)	Finland	Coal	1.07-18.15	Damage cost (bottom-up)
European Commission (1999)	France	Coal	9.61-29.45	Damage cost (bottom-up)
European Commission (1999)	Greece	Oil	2.07-19.89	Damage cost (bottom-up)
European Commission (1999)	Germany	Coal	2.38-23.67	Damage cost (bottom-up)
		Oil	5.30-35.16	
		Gas	0.83-9.55	
European Commission (1999)	Ireland	Coal	6.16-31.90	Damage cost (bottom-up)
European Commission (1999)	Italy	Oil	3.24-24.52	Damage cost (bottom-up)
		Gas	1.21-11.78	
European Commission (1999)	The Netherlands	Coal	1.68-24.48	Damage cost (bottom-up)
		Gas	0.43-9.65	
European Commission (1999)	Portugal	Coal	3.69-30.22	Damage cost (bottom-up)
European Commission (1999)	Spain	Coal	4.64-32.60	Damage cost (bottom-up)
European Commission (1999)	Sweden	Coal	0.84-16.93	Damage cost (bottom-up)
European Commission (1999)	UK	Coal	4.06-33.01	
		Oil	3.22-22.10	
		Gas	0.73-10.21	
Hirschberg and Jakob (1999)	Switzerland	Coal	4.54-23.16	Damage cost (bottom-up)
		Oil	5.13-26.09	
		Gas	1.17-8.06	
Maddison (1999)	UK/Germany	Coal	0.31/0.71	Damage cost (bottom-up)
		Gas	0.78	

DRF = dose response function, NO2= nitrogen dioxide, PM10= Particulate matter up to 10 micrometers in size, RHA = Respiratory Hospital Admission, SO2= sulfur dioxide, WLD = Work loss days.

Source: *Soderholm, P. and T. Sundqvist. 2006. Measuring environmental externalities in the electric power sector. Environmental Valuation in Developed Countries, Case Studies. 8: 148-179.*

APPENDIX 6: SENSITIVITY ANALYSIS

Appendix Table 6.1: Sensitivity Analysis for High Level of Abatement (95%–85%–99% for SO_2, NO_2, and PM_{10})

	Mortality valuation					
	Rs. (million)			$ (million)		
Morbidity valuation	Lower DRF	Average DRF	Upper DRF	Lower DRF	Averag e DRF	Upper DRF
Lower DRF						
COI (100% public RHA + WLD)						
S.F. = 1.65	885.00	1,517.83	2,147.45	19.60	33.61	47.55
S.F. = 2.14	920.26	1,553.09	2,182.71	20.38	34.39	48.33
S.F. = 3.16	988.02	1,620.85	2,250.47	21.88	35.89	49.83
COI (25% public & 75% private RHA + WLD)						
S.F. = 1.65	992.34	1,625.17	2,254.79	21.97	35.99	49.93
S.F. = 2.14	1,060.78	1,693.62	2,323.23	23.49	37.50	51.44
S.F. = 3.16	1,192.30	1,825.13	2,454.75	26.40	40.41	54.36
COI (50% public & 50% private RHA + WLD)						
S.F. = 1.65	956.56	1,589.39	2,219.01	21.18	35.19	49.14
S.F. = 2.14	1,013.94	1,646.78	2,276.39	22.45	36.47	50.41
S.F. = 3.16	1,124.21	1,757.04	2,386.66	24.89	38.91	52.85
Average DRF						
COI (100% public RHA + WLD)						
S.F. = 1.65	1,037.97	1,670.81	2,300.42	22.98	37.00	50.94
S.F. = 2.14	1,120.52	1,753.35	2,382.97	24.81	38.83	52.77
S.F. = 3.16	1,279.14	1,911.97	2,541.59	28.32	42.34	56.28
COI (25% public & 75% private RHA + WLD)						
S.F. = 1.65	1,292.60	1,925.44	2,555.05	28.62	42.64	56.58
S.F. = 2.14	1,453.85	2,086.69	2,716.30	32.19	46.21	60.15
S.F. = 3.16	1,763.71	2,396.54	3,026.16	39.05	53.07	67.01
COI (50% public & 50% private RHA + WLD)						
S.F. = 1.65	1,207.73	1,840.56	2,470.18	26.74	40.76	54.70
S.F. = 2.14	1,342.74	1,975.58	2,605.19	29.73	43.75	57.69
S.F. = 3.16	1,602.19	2,235.02	2,864.64	35.48	49.49	63.43
Upper DRF						
COI (100% public RHA + WLD)						
S.F. = 1.65	1,266.56	1,899.39	2,529.01	28.05	42.06	56.00
S.F. = 2.14	1,419.76	2,052.59	2,682.21	31.44	45.45	59.39
S.F. = 3.16	1,714.14	2,346.98	2,976.59	37.96	51.97	65.91
COI (25% public & 75% private RHA + WLD)						
S.F. = 1.65	1,740.92	2,373.75	3,003.37	38.55	52.56	66.51
S.F. = 2.14	2,040.74	2,673.57	3,303.19	45.19	59.20	73.14
S.F. = 3.16	2,616.86	3,249.70	3,879.31	57.95	71.96	85.90
COI (50% public & 50% private RHA + WLD)						
S.F. = 1.65	1,582.80	2,215.63	2,845.25	35.05	49.06	63.00
S.F. = 2.14	1,833.74	2,466.58	3,096.20	40.61	54.62	68.56
S.F. = 3.16	2,315.96	2,948.79	3,578.41	51.28	65.30	79.24

Source: Authors' estimates.
Notes: Default valuation estimates are highlighted.
Each cell corresponds to total health cost, as a summation of morbidity and mortality.
$1 = Rs. 45.16; S.F. = WTP/COI.

Appendix Table 6.2: Sensitivity Analysis for Average Level of Abatement (48%–43%–50% for SO_2, NO_2, and PM_{10})

	Mortality valuation					
	Rs. (million)			$ (million)		
Morbidity valuation	Lower DRF	Average DRF	Upper DRF	Lower DRF	Average DRF	Upper DRF
Lower DRF						
COI (100% public RHA + WLD)						
S.F. = 1.65	6,383.49	11,327.66	16,233.49	141.35	250.83	359.47
S.F. = 2.14	6,786.92	11,731.10	16,636.93	150.29	259.77	368.40
S.F. = 3.16	7,562.16	12,506.34	17,412.17	167.45	276.93	385.57
COI (25% public & 75% private RHA + WLD)						
S.F. = 1.65	7,507.39	12,451.56	17,357.40	166.24	275.72	384.35
S.F. = 2.14	8,258.22	13,202.39	18,108.22	182.87	292.35	400.98
S.F. = 3.16	9,700.98	14,645.16	19,550.99	214.81	324.29	432.93
COI (50% public & 50% private RHA + WLD)						
S.F. = 1.65	7,132.75	12,076.93	16,982.76	157.94	267.43	376.06
S.F. = 2.14	7,767.79	12,711.96	17,617.79	172.01	281.49	390.12
S.F. = 3.16	8,988.04	13,932.22	18,838.05	199.03	308.51	417.14
Average DRF						
COI (100% public RHA + WLD)						
S.F. = 1.65	8,243.67	13,187.84	18,093.67	182.54	292.02	400.66
S.F. = 2.14	9,222.07	14,166.25	19,072.08	204.21	313.69	422.32
S.F. = 3.16	11,102.14	16,046.32	20,952.15	245.84	355.32	463.95
COI (25% public & 75% private RHA + WLD)						
S.F. = 1.65	11,149.70	16,093.88	20,999.71	246.89	356.37	465.01
S.F. = 2.14	13,026.34	17,970.51	22,876.34	288.45	397.93	506.56
S.F. = 3.16	16,632.42	21,576.59	26,482.42	368.30	477.78	586.41
COI (50% public & 50% private RHA + WLD)						
S.F. = 1.65	10,181.02	15,125.20	20,031.03	225.44	334.92	443.56
S.F. = 2.14	11,758.25	16,702.42	21,608.26	260.37	369.85	478.48
S.F. = 3.16	14,788.99	19,733.17	24,639.00	327.48	436.96	545.59
Upper DRF						
COI (100% public RHA + WLD)						
S.F. = 1.65	10,664.51	15,608.68	20,514.52	236.15	345.63	454.26
S.F. = 2.14	12,391.17	17,335.35	22,241.18	274.38	383.87	492.50
S.F. = 3.16	15,709.08	20,653.25	25,559.09	347.85	457.34	565.97
COI (25% public & 75% private RHA + WLD)						
S.F. = 1.65	15,870.09	20,814.26	25,720.09	351.42	460.90	569.53
S.F. = 2.14	19,205.75	24,149.92	29,055.75	425.28	534.76	643.40
S.F. = 3.16	25,615.45	30,559.62	35,465.45	567.22	676.70	785.33
COI (50% public & 50% private RHA + WLD)						
S.F. = 1.65	14,134.89	19,079.07	23,984.90	313.00	422.48	531.11
S.F. = 2.14	16,934.22	21,878.40	26,784.23	374.98	484.46	593.10
S.F. = 3.16	22,313.32	27,257.50	32,163.33	494.09	603.58	712.21

Source: *Authors' estimates.*

Appendix Table 6.3: Sensitivity Analysis for Low Level of Abatement (24%–21%–25% for SO_2, NO_2, and PM_{10})

	Mortality valuation					
	Rs. (million)			$ (million)		
Morbidity valuation	Lower DRF	Average DRF	Upper DRF	Lower DRF	Average DRF	Upper DRF
Lower DRF						
COI (100% public RHA + WLD)						
S.F. = 1.65	9,132.73	16,232.58	23,276.52	202.23	359.45	515.42
S.F. = 2.14	9,720.26	16,820.10	23,864.04	215.24	372.46	528.43
S.F. = 3.16	10,849.23	17,949.08	24,993.02	240.24	397.46	553.43
COI (25% public & 75% private RHA + WLD)						
S.F. = 1.65	10,764.91	17,864.76	24,908.70	238.37	395.59	551.57
S.F. = 2.14	11,856.93	18,956.78	26,000.72	262.55	419.77	575.75
S.F. = 3.16	13,955.32	21,055.17	28,099.11	309.02	466.23	622.21
COI (50% public & 50% private RHA + WLD)						
S.F. = 1.65	10,220.85	17,320.70	24,364.64	226.33	383.54	539.52
S.F. = 2.14	11,144.71	18,244.55	25,288.49	246.78	404.00	559.98
S.F. = 3.16	12,919.96	20,019.80	27,063.74	286.09	443.31	599.29
Average DRF						
COI (100% public RHA + WLD)						
S.F. = 1.65	11,846.51	18,946.36	25,990.30	262.32	419.54	575.52
S.F. = 2.14	13,272.85	20,372.69	27,416.63	293.91	451.12	607.10
S.F. = 3.16	16,013.64	23,113.49	30,157.43	354.60	511.81	667.79
COI (25% public & 75% private RHA + WLD)						
S.F. = 1.65	16,078.25	23,178.10	30,222.04	356.03	513.24	669.22
S.F. = 2.14	18,812.58	25,912.42	32,956.36	416.58	573.79	729.77
S.F. = 3.16	24,066.77	31,166.62	38,210.56	532.92	690.14	846.12
COI (50% public & 50% private RHA + WLD)						
S.F. = 1.65	14,667.67	21,767.52	28,811.46	324.79	482.01	637.99
S.F. = 2.14	16,966.00	24,065.85	31,109.79	375.69	532.90	688.88
S.F. = 3.16	21,382.40	28,482.24	35,526.18	473.48	630.70	786.67
Upper DRF						
COI (100% public RHA + WLD)						
S.F. = 1.65	15,363.49	22,463.33	29,507.27	340.20	497.42	653.39
S.F. = 2.14	17,876.88	24,976.73	32,020.67	395.86	553.07	709.05
S.F. = 3.16	22,706.55	29,806.39	36,850.33	502.80	660.02	815.99
COI (25% public & 75% private RHA + WLD)						
S.F. = 1.65	22,934.67	30,034.52	37,078.46	507.85	665.07	821.05
S.F. = 2.14	27,788.25	34,888.10	41,932.04	615.33	772.54	928.52
S.F. = 3.16	37,114.74	44,214.59	51,258.53	821.85	979.07	1,135.04
COI (50% public & 50% private RHA + WLD)						
S.F. = 1.65	20,410.94	27,510.79	34,554.73	451.97	609.18	765.16
S.F. = 2.14	24,484.46	31,584.31	38,628.25	542.17	699.39	855.36
S.F. = 3.16	32,312.01	39,411.85	46,455.79	715.50	872.72	1,028.69

Source: Authors' estimates.

Appendix Table 6.4: Mortality and Morbidity Incidence, by Abatement Level

Health Impact by Abatement Level	Lower bound DRF	Average DRF	Upper bound DRF
Low Abatement			
Mortality	339	668	995
Morbidity			
RHA	100,912	260,923	467,392
SO_2	62,868	26,719	62,344
NO_2	12,019	65,044	118,775
PM_{10}	26,025	169,161	286,272
WLD	377,359	409,890	507,483
Average Abatement			
Mortality	482	339	995
Morbidity			
RHA	146,548	379,954	679,792
SO_2	91,308	38,806	90,547
NO_2	16,461	89,082	162,671
PM_{10}	38,779	252,067	426,574
WLD	562,302	610,777	756,200
High Abatement			
Mortality	51	94	136
Morbidity			
RHA	9,638	22,862	42,591
SO_2	5,987	2,545	5,938
NO_2	3,135	16,968	30,985
PM_{10}	515	3,350	5,669
WLD	7,472	8,117	10,049

DRF = dose response function, NO_2= nitrogen dioxide, PM_{10}= Particulate matter up to 10 micro meters in size, RHA = Respiratory Hospital Admission, SO_2= sulfur dioxide, WLD = Work loss days.

Source: Authors' estimates.

Note: Default scenario is highlighted.

Valuation of Health Impacts of Air Pollution from Power Plants in Asia
A Practical Guide

Assigning a monetary value for air quality reduction and associated health outcomes of electricity generation is both difficult and essential. From a practical point of view, conducting complete and detailed studies for every power plant project is not feasible. This paper reviews the Impact Pathway Approach for valuing health costs of air pollution and recommends a streamlined methodology combining site-specific studies and benefit transfer for quick assessments. Strengthening available regulatory measures of pollution control and implementing a rigorous monitoring program to ensure installation and use of pollution control equipment are therefore welfare improving.

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