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Health damages from air pollution in China

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ABSTRACT

This study evaluates air pollution-related health impacts on the Chinese economy by using an expanded version of the Emissions Prediction and Policy Analysis model. We estimated that marginal welfare impact to the Chinese economy of ozone and particulate-matter concentrations above background levels increased from 1997 US$22 billion in 1975 to 1997 US$112 billion in 2005, despite improvements in overall air quality. This increase is a result of the growing urban population and rising wages that thus increased the value of lost labor and leisure. In relative terms, however, welfare losses from air pollution decreased from 14% of the historical welfare level to 5% during the same period because the total size of the economy grew much faster than the absolute air pollution damages. In addition, we estimated that particulate-matter pollution alone led to a gross domestic product loss of 1997 US$64 billion in 1995. Given that the World Bank's comparable estimate drawn from a static approach was only 1997 US$34 billion, this result suggests that conventional static methods neglecting the cumulative impact of pollution-caused welfare damage are likely to underestimate pollution-health costs substantially. However, our analysis of uncertainty involved in exposure–response functions suggests that our central estimates are susceptible to significantly large error bars of around ±80%.

1. Introduction

As China continues a three decade-long trajectory of unprecedented growth and development, there has been increasing concern that its growth has come at substantial cost to its environment and public health. While there have been important quality of life improvements for the more than half a billion people who have been raised out of poverty, most traditional accounting has not fully considered the impacts of health and environmental externalities in analysis of China's development. China now faces severe challenges relating to its environment, including air pollution, the availability of clean water, and desertification. Issues such as these have the potential to create constraints on future growth. Those environmental problems that result in negative health outcomes, such as contaminated water and high levels of air pollution, also incur real costs on the individuals, the health system, and the economy as a whole.

Many studies have attempted to quantify the economic costs of air pollution (e.g., Ostro and Chestnut, 1998; EPA, 1999; Holland et al., 2005; Saikawa et al., 2009; Vennemo et al., 2006; West et al., 2006). One serious challenge faced in estimating these pollution costs has been associating dollar values with such "non-market" impacts as lost lives, biodiversity loss, and landscape degradation. Studies focusing on the negative health consequences associated with air pollution have the advantage of dealing at least partially with outcomes that involve economic transactions, such as payment for health services or the loss of labor and leisure time to combat illness. For this reason, they are a useful first step in the larger process of determining how to integrate environmental externalities into larger economic analyses.

In the case of China, there have been several studies, at both the local and national levels, that have worked to quantify the economic costs of air pollution arising from its negative impact on human health (Nielsen and Ho, 2007). Most of them (e.g., Aunan et al., 2004; Hirschberg et al., 2003; O'Connor et al., 2003; World Bank and SEPA, 2007) draw aggregate damage functions, and apply them to the target air quality level in static ways to estimate associated health damage. Such aggregate damage function approaches or point estimates, however, may not fully reflect the overall economic impacts because they do not explicitly identify how resources and goods/service demands are affected by pollution.

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In this study, we aim to improve the conventional approach in several key aspects and offer an estimate of long-term economic impacts caused by air pollution in China. We incorporate health-related environmental damages into an integrated assessment model that combines broader socio-economic aspects of air pollution with scientific models of atmospheric chemistry, urban air pollution, ocean, and terrestrial systems. Our integrated assessment method explicitly describes how the supply and demand of resources and of goods and services are affected by pollution, and by capturing the changes in demands for goods and services throughout the economy, we are able to draw a robust picture of how changes in pollution, and their associated health impacts, have historically affected the Chinese economy.

2. Theoretical framework and method: EPPA–HE

We use the fourth version of the MIT Emissions Prediction Policy Analysis (EPPA) model (Paltsev et al., 2005), which is a multi-region,[2] multi-sector computable general equilibrium (CGE) model of the world economy built on the Global Trade Analysis Project 5 (GTAP5) dataset.[3] EPPA version 4 (EPPA4) can be easily modified to include valuation of health impacts. In particular, the following modifications are made to EPPA4 to estimate historic health impacts of air pollution in China. We start our data and analysis from 1970, and include the household healthcare production and leisure in the social accounting matrix. Introduction of a household healthcare production sector that provides "pollution health services" allows us to capture the health effects related to both morbidity and mortality. Our model also calculates the incidence and overall costs of each health outcome, such as restricted activity days, respiratory hospital admissions, asthma attacks, and other morbidity and mortality outcomes from acute and chronic exposure. The model calculates the service, labor and leisure costs of all impact categories (often referred to as "health endpoints" in epidemiological literature). We call the modified model EPPA–HE (EPPA–Health Effects).

In each time period between 1970 and 2005, for each pollutant, the model calculates the number of cases of every health outcome, given a pollution level and the number of people exposed to each pollutant. Once the number of cases is computed, the model then calculates corresponding costs, determined by health service inputs, lost labor, and leisure time needed to deal with the illness. It also calculates changes in the quantity of the service input, labor, and leisure. The totals for all endpoints and pollutants are passed into the economic system as a change in the total amount of labor supply available. The total labor supply and changes in it are allocated between labor and leisure depending on the specification of the labor supply elasticity and changes in the endogenously modeled wage rate. Greater (or fewer) medical service needs are treated as a reduction (increase) in the productivity of the household healthcare sector. Introducing these changes as shocks into the general equilibrium model makes it possible to capture the interactions that occur among the different sectors of the economy. The details of the calculations and their sensitivities used in EPPA–HE are identical to those described by Matus et al. (2008), Selin et al. (2009), and Nam et al. (2010).

In order to properly account for the effects of mortality from chronic exposure, which prematurely removes workers from the workforce, the model keeps track of the lifetime exposure of each age cohort. The premature deaths from chronic exposure have an effect beyond the immediate time period in which they occur. When an individual dies at 40 years of age, assuming that his or her retirement age is 65, then the economy loses 25 years worth of labor from this individual. EPPA-HE is able to track deaths in each period, and propagate them forward until the point where they no longer represent a loss to the economy (the year in which the individual would no longer have been part of the workforce). It also performs a similar calculation for the amount of leisure lost, assuming that the individual would have only leisure time, not wage income, in the period after they left the workforce and before they died.

In order to get the full effect of past mortalities as a loss of available labor in the economy, the model takes the sum of mortalities from chronic exposure for all previous years. This requires not just an accounting of total premature deaths, but also a calculation of how much labor is lost for each of those deaths. Because labor productivity increases over time, the value of lost labor also increases over time. So if an individual dies five years before they would have left the workforce, the economy loses not only those five years of labor, but also extra labor that would have been available due to productivity increases. To consider changes in labor productivity, each past death is multiplied by the changes in labor productivity for each year since the death. Once the overall labor lost due to premature death is computed, then the total sum of deaths across all cohorts is subtracted from available labor in the main model.

For EPPA–HE, the Chinese economy and population has also been historically benchmarked in the same manner as done for the United States and Europe (Matus et al., 2008; Nam et al., 2010). Our model investigates the effects of two pollutants—ozone (O_3), and particulates of 10 microns or less (PM_{10})—and includes demographic and pollution data specific to China.

3. Air quality data

3.1. Historic concentrations of fine particulates

Fine particulates in the air, which cause respiratory and cardiovascular diseases, are one of the key pollutants that account for a large fraction of damage on human health (EPA, 1997). The World Health Organization (WHO) and many national level public health agencies have adopted fine particles that are smaller than 2.5 micrometers ($PM_{2.5}$) or 10 micrometers (PM_{10}) in terms of diameter as key metrics to control PM levels (Holland et al., 1999). In particular, $PM_{2.5}$ is known to be a better predictor for PM-driven acute and chronic health effects than coarse mass (Schwartz et al., 1996; Cifuentes et al., 2000). This paper, however, focuses on PM_{10} rather than on $PM_{2.5}$ due to data availability.

China's Ministry of Environmental Protection (MEP)—formerly, the State Environmental Protection Administration (SEPA)—has monitored PM levels (in terms of total suspended particulate concentrations) in major Chinese cities on a regular basis since the early 1980s. As illustrated in Fig. 1, PM concentration levels in China vary by location. On average, northern cities show much higher PM levels than southern cities. In this sense, it is crucial to deal with this spatial variation in PM concentrations to come up with a reasonable national number because EPPA-HE is designed to apply one national-level air quality index for each year to the affected population group. For this matter, we first chose 34 major Chinese cities,[4] for which China's official PM concentration data are relatively complete for the last three decades and which proportionally represent China's northern and southern regions. Then, we computed their population-weighted average for each year. To consider the possibility that our estimates for PM concentrations are somewhat upward biased compared to China's

[2] EPPA disaggregates the whole world into 16 regions, and China represents one of them.

[3] As GTAP5 is a snapshot of the 1997 world economy, EPPA4 rebases its key economic inputs to 1997.

[4] The list of these 34 cities is the same as the one displayed in Fig. 1.

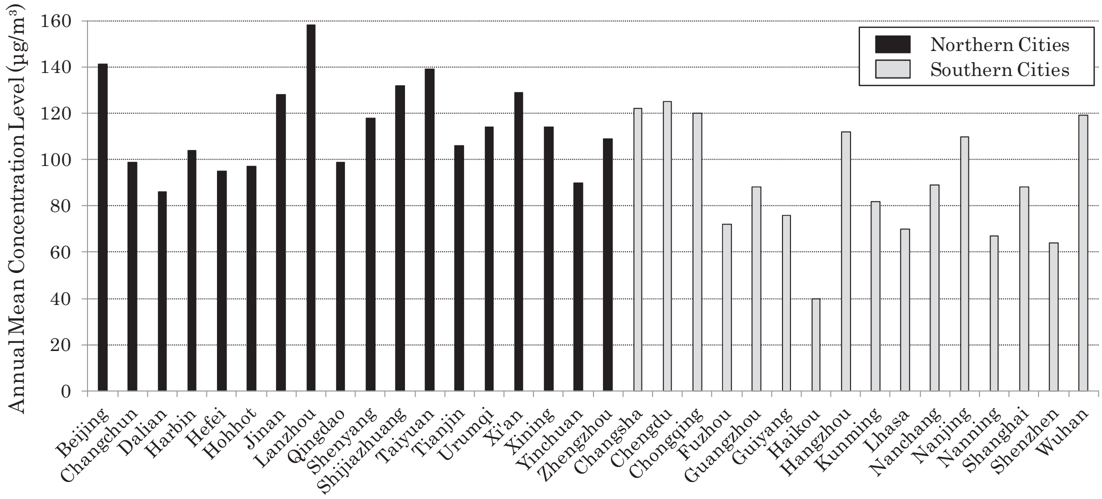

Fig. 1. PM_{10} concentration levels in 34 major Chinese cities, 2005.
Source: Data from SEPA (2006).

actual national average numbers (as our estimates exclude PM concentration levels in rural China), we apply the PM levels only to urban population.

Another issue to deal with in estimating China's historic PM concentration levels is the conversion between total suspended particulate (TSP) and PM_{10}. SEPA used TSP as a primary monitoring metric of fine particulates until 2002, and changed it into PM_{10} in 2003. As most epidemiological studies focus on PM_{10} (or $PM_{2.5}$) to draw exposure–response functions, we convert TSP concentration levels into PM_{10} measures so that we can incorporate a broad range of the epidemiological literature into our study. Most studies focusing on China's air pollution use 0.5–0.65 as TSP–PM_{10} conversion factors (Table 1). Among them, we choose the smallest conversion factor (0.5) to compute our central estimates for PM-caused health damage. However, as shown in Fig. 2, our PM estimates for 1981–2002 may underestimate PM_{10} levels by up to 30% compared with those based on the conversion factor of 0.65. To quantify the impact of the conversion factor on our impact estimates, we conduct sensitivity analysis with regard to TSP–PM_{10} conversion factors in Section 6.

3.2. Historic concentrations of ozone

Although ozone is a crucial pollutant that causes serious damage to human health, China only recently began monitoring ozone levels. For our analysis period of 1970–2005, official measured data on ozone concentration in China do not exist. For this reason, most studies analyzing air pollution in China have excluded ozone from their analysis (e.g., World Bank, 1997; Ho et al., 2002; Aunan et al., 2004; World Bank and SEPA, 2007).

We estimate Chinese ozone concentration from modeled data. To generate this data, we first adopt $1^\circ \times 1.25^\circ$ global afternoon

Table 1
TSP-PM_{10} conversion factors used for studies on China's air pollution.

World Bank (1997)	Lvovsky et al. (2000)*	Aunan and Pan (2004)	Wan (2005)	Levy and Greco (2007)	World Bank and SEPA (2007)
0.6	0.65	0.6	0.55	0.54	0.5

* Lvovsky et al. (2000) focus on Shanghai and five other non-Chinese cities including Mumbai, Manila, Bangkok, Krakow, and Santiago.

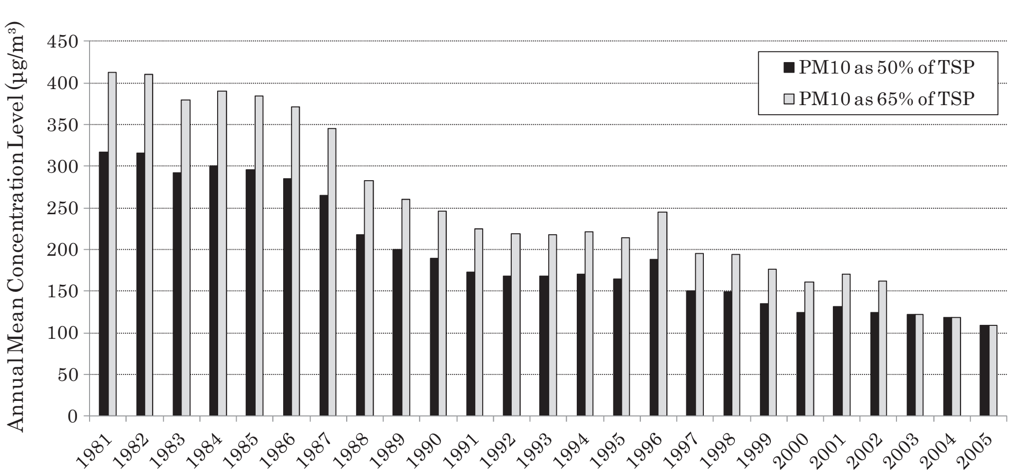

Fig. 2. PM_{10} concentration levels in China, 1981–2005.
Source: Computed from World Bank (2001), State Environmental Protection Administration (SEPA) (1997–2008), and NBSC (1982–2006).

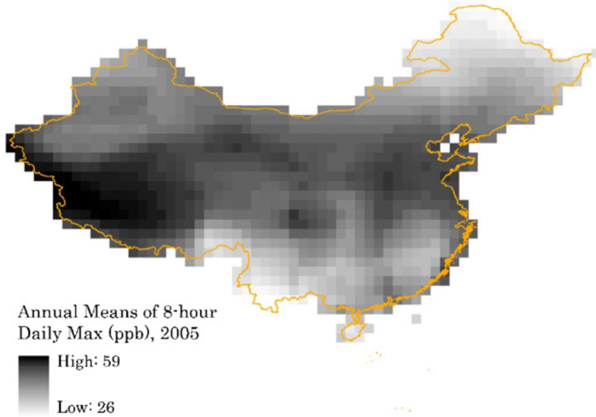

Fig. 3. Ozone concentration levels in China by 1° × 1° Grid Cell, 2005. *Source*: Converted from original 1° × 1.25° data (Lamsal et al., 2011).

ozone concentration simulation data from the GEOS-Chem model (Lamsal et al., 2011). GEOS-Chem is a global three-dimensional chemical transport model for atmospheric composition, which is built on meteorological input from the Goddard Earth Observing System of the NASA Global Modeling and Assimilation Office (Bey et al., 2001). Annual mean afternoon ozone, simulated by GEOS-Chem, is a metric comparable to annual mean 8-hour daily maximum ozone (Selin et al., 2009). We convert this original 1°×1.25° data into 1°×1° data by using the inverse distance weighted spatial interpolation method (Fig. 3). As GEOS-Chem does not consider ozone titration by nitrogen oxides, which may occur around large urban areas, there is a possibility that the ozone levels simulated by the model may be somewhat overestimated.

For historic ozone levels, we use zonal means of ozone concentration for 1970–2000 simulated by the Integrated Global System Model (IGSM) climate simulation (See Sokolov et al. (2005), for details on the IGSM). For years prior to 2005, we scale 2005 concentrations based on zonal means from the IGSM and create 1°×1° ozone concentration maps. Finally, we calculate population-weighted average ozone concentration for each year by applying a 1°×1° population grid map (SEDAC, 2009) (Fig. 4). Thus, for ozone (in contrast to our methodology for PM), China's entire population is affected by these pollutants.

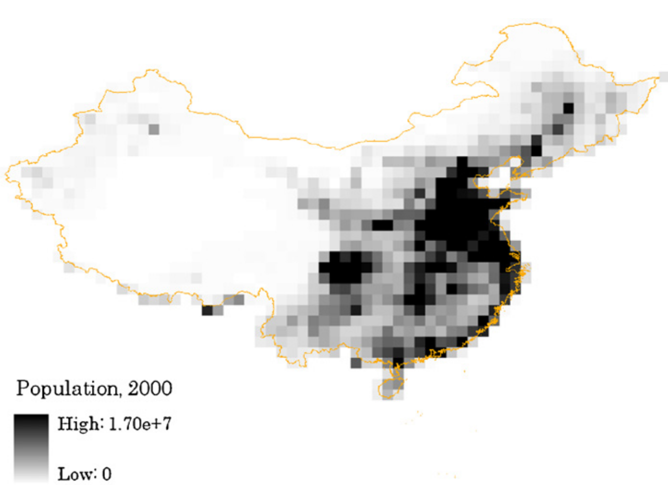

Fig. 4. China's population by 1°×1° Grid Cell, 2000. *Source*: Data from SEDAC (2009).

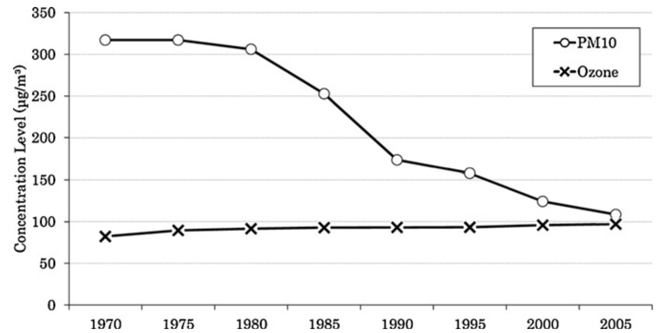

Fig. 5. PM_{10} and O_3 concentration levels used in EPPA-HE, 1970–2005.

3.3. Air quality input for EPPA-HE

Fig. 5 illustrates PM and O_3 concentration levels used as input for EPPA-HE. In the case of PM, we use five-year average for each year (e.g., the number for 1985 is an average concentration level for the period of 1985–1989), as EPPA-HE simulates socio-economic projections for each 5-year interval. But in the case of O_3, each year's concentration level in the figure is the one computed from each year's grid map. PM_{10} concentration levels for 1970 and 1975, for which China's official PM data do not exist, are assumed to be comparable to those for 1981, when SEPA began systemic monitoring on TSP.

4. Case computation and valuation

In this section, we briefly describe the health endpoint and valuation module of EPPA-HE. As previously mentioned, the methodology we adopted for this study is identical to the ones previously applied to the US and Europe. Here, we present updates to the methodology and China-specific parameters.

4.1. Health endpoints and exposure–response functions

EPPA-HE links air quality and associated health outcomes by using exposure–response (ER) functions. ER functions quantify how many health-end outcomes or change in the death rate caused by a unit increase in a pollutant's concentration level. Epidemiologists (e.g., Anderson et al., 2004; Aunan and Pan, 2004; Dockery et al., 1993; Hiltermann et al., 1998; Hurley et al., 2005; Künzli et al., 2000; Ostro and Rothschild, 1989; Pope et al., 1995, 2002; Pope et al., 2004; Samet et al., 2000; Venners et al., 2003; Zhang et al., 2002) have undertaken a number of research projects to establish reliable ER functions, and the ExternE project (Holland et al., 1999; Bickel and Friedrich, 2005), initiated by the European Commission, synthesizes existing epidemiological studies and provides a comprehensive list of well-established ER relationships. Our study adopts ER functions from two ExternE studies. We put priority on updated ER functions, recommended by Bickel and Friedrich (2005), but also use ER functions from Holland et al. (1999) when updated ER functions do not exist. All of the ER functions used for our study are listed in Table 2.

All ER functions in our study take a linear form and do not assume any threshold effects. In particular, we compute the number of cases of non-fatal health outcomes, caused by air pollution, with the following equation:

$$\text{Cases}_{ijt}^{\text{Morbidity}} = \text{ER}_{ij} \cdot C_{jt} \cdot P_t \quad (1)$$

where ER_{ij}, C_{jt}, and P_t refer to ER function for health-end outcome i and pollutant j, concentration level of pollutant j at time t, and

Table 2
Exposure–response functions.

Receptor	Impact category		ER function[a]	C.I. (95%) Low	C.I. (95%) High	Computed or adapted from[b]
Entire age groups	Respiratory hospital admissions	PM_{10}	7.03E−06	3.83E−06	1.03E−05	ExternE (2005)
		O_3	3.54E−06	6.12E−07	6.47E−06	ExternE (1999)
	Cerebrovascular hospital admissions	PM_{10}	5.04E−06	3.88E−07	9.69E−06	ExternE (2005)
	Cardiovascular hospital admissions	PM_{10}	4.34E−06	2.17E−06	6.51E−06	ExternE (2005)
	Respiratory symptoms days	O_3	3.30E−02	5.71E−03	6.03E−02	ExternE (1999)
	Asthma attacks	O_3	4.29E−03	3.30E−04	8.25E−03	ExternE (1999)
	Mortality from Acute Exposure	O_3	0.03%	0.01%	0.04%	ExternE (2005)
		PM_{10}	0.06%	0.04%	0.08%	ExternE (2005)
	Mortality from Chronic Exposure	PM_{10}	0.25%	0.02%	0.48%	Pope et al. (2002)
Children	Chronic Bronchitis	PM_{10}	1.61E−03	1.24E−04	3.10E−03	ExternE (1999)
	Chronic Cough	PM_{10}	2.07E−03	1.59E−04	3.98E−03	ExternE (1999)
	Respiratory symptoms days	PM_{10}	1.86E−01	9.20E−02	2.77E−01	ExternE (2005)
	Bronchodilator usage	PM_{10}	1.80E−02[c]	−6.90E−02	1.06E−01	ExternE (2005)
	Cough	O_3	9.30E−02[d]	−1.90E−02	2.22E−01	ExternE (2005)
	Lower respiratory symptoms (wheeze)	PM_{10}	1.86E−01[e]	9.20E−02	2.77E−01	ExternE (2005)
		O_3	1.60E−02[f]	−4.30E−02	8.10E−02	ExternE (2005)
Adults	Restricted activity day	PM_{10}	5.41E−02[g]	4.75E−02	6.08E−02	ExternE (2005)
	Minor restricted activity day	O_3	1.15E−02[h]	4.40E−03	1.86E−02	ExternE (2005)
		PM_{10}	3.46E−02[h]	2.81E−02	4.12E−02	ExternE (2005)
	Work loss day	PM_{10}	1.24E−02[h]	1.06E−02	1.42E−02	ExternE (2005)
	Respiratory symptoms days	PM_{10}	1.30E−01[i]	1.50E−02	2.43E−01	ExternE (2005)
	Chronic bronchitis	PM_{10}	2.65E−05	−1.90E−06	5.41E−05	ExternE (2005)
	Bronchodilator usage	PM_{10}	9.12E−02[j]	−9.12E−02	2.77E−01	ExternE (2005)
		O_3	7.30E−02[j]	−2.55E−02	1.57E−01	ExternE (2005)
	Lower respiratory symptoms (wheeze)	PM_{10}	1.30E−01[k]	1.50E−02	2.43E−01	ExternE (2005)
Elderly 65+	Respiratory hospital admissions	O_3	1.25E−05	−5.00E−06	3.00E−05	ExternE (2005)
	Congestive heart failure	PM_{10}	1.85E−05	1.42E−06	3.56E−05	ExternE (1999)

Source: Modified from Nam et al. (2010).

[a] E–R functions for mortality from acute and chronic exposure have the unit of % increase in annual mortality rate/(μg/m^3), while the other E–R functions are measured in cases/(yr-person-μg/m^3).
[b] ExternE (1999) and ExternE (2005) refer to Holland et al. (1999) and Bickel and Friedrich (2005), respectively.
[c] Defined on children aged 5–14 years meeting certain criteria (around 15–25% of child population).
[d] ER functions on cough for O_3 are defined on general population of ages 5–14.
[e] LRS values for PM_{10} include impacts on cough.
[f] LRS ER functions for O_3 are defined on general population of ages 5–14.
[g] Restricted activity days include both minor restricted days and work loss days.
[h] Part of restricted activity days.
[i] Defined on adults population with chronic respiratory symptoms (around 30% of adult population).
[j] Defined on population of >20 with well-established asthma (around 4.5% of total adult population).
[k] LRS ER functions for PM are defined on adult population with chronic respiratory symptoms (around 30% of total adult population); ExternE (2005) LRS values for PM include impacts on cough.

affected population group at time t, respectively. Similarly, we compute the number of premature deaths from acute exposure by using the equation:

$$\text{Cases}_t^{\text{AM}} = \sum_j \text{ER}_j^{\text{AM}} \cdot C_{jt} \cdot M_t \cdot P_t \qquad (2)$$

where ER_j^{AM} and M_t refer to ER function for mortality from acute exposure and pollutant j and overall mortality rate at time t. These numbers of cases are then valuated in terms of year 1997 US$ by using unit values displayed in Table 3.[5] We follow the approach of others (Bickel and Friedrich, 2005) and assume that life lost due to acute exposure is 6 months on average, as primary victims susceptible to premature deaths associated with acute exposure are those who already have impaired health conditions for other reasons.

[5] There is a concern that cohort studies cannot fully distinguish mortalities associated with acute exposure from those brought forward by chronic exposure, and thus an ER function for mortalities from chronic exposure may include part of the acute exposure effect (Bickel and Friedrich, 2005). If this is the case, then the application of an ER function for acute exposure may result in somewhat overestimated mortality effects. However, as shown in Table 9, the direct loss due to mortalities from acute exposure accounts for a very tiny portion of the total welfare loss (0.3% in 2005), and thus the degree of overestimation would be marginal.

Table 3
Valuation of health endpoints in China.

Outcome	Unit	Cost (1997 US$)
Hospital admission[a]	per admission	284
Emergency room visits for respiratory illness[a]	per visit	23
General practitioner visits		
Asthma[a]	per consultation	4
Lower respiratory symptoms[a]	per consultation	13
Respiratory symptoms in asthmatics[a]	per event	0.60
Respiratory medication use	per day	0
Restricted activity day	per day	2.32
Cough day	per day	0.60
Symptom day	per day	0.60
Work loss day	per day	1.43
Minor restricted activity day	per day	0.60
Chronic bronchitis[a]	per case	8,000
Mortality from acute exposure	per case	662

Note: All values displayed in this table are estimated by willingness-to-pay surveys and market data.
Source: [a]As estimated for China by World Bank and SEPA (1997: 25); For other endpoints, we adjusted the European valuation table presented in Bickel and Friedrich (2005: 156) by using the average cost difference between the valuation table for China estimated by World Bank and SEPA (1997: 25) and that for Europe estimated by Bickel and Friedrich (2005: 156).

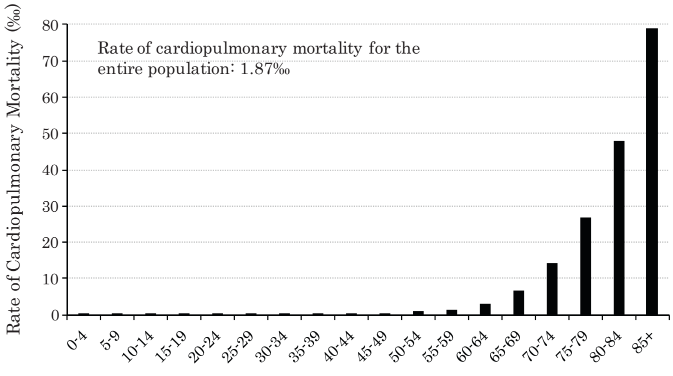

Fig. 6. Rate of cardiopulmonary mortality in China, 2003.
Source: Data from Ministry of Health, People's Republic of China (2004).

4.2. Age-conditioned ER functions for mortality from chronic exposure

In this study, we deal with premature deaths from chronic exposure to PM in a different way from other studies.[6] A conventional approach to valuing mortality from chronic exposure is to apply a constant ER function, such as 0.25% (as displayed in Table 2) to the entire population group. This approach does not reflect the exposure to PM accumulated over a lifetime. In other words, chronic exposure occurs over time and thus should be dealt with as a function of age. Rates of death from heart and lung diseases, which comprise the majority of premature deaths caused by chronic exposure to excess PM concentrations (Holland et al., 1999), are substantially higher for older population groups in China (Fig. 6).

In particular, we calculate an age-conditioned ER function for mortality from chronic exposure by adjusting the unconditioned ER function for mortality from chronic exposure displayed in Table 2 using the following equation:

$$\mathrm{ER}_n^{\mathrm{CM}} = \mathrm{ER}^{\mathrm{CM}} \cdot \frac{M_n^{\mathrm{CPL}}/M_n^{\mathrm{All}}}{M^{\mathrm{CPL}}/M^{\mathrm{All}}} \tag{3}$$

where $\mathrm{ER}^{\mathrm{CM}}$ and $\mathrm{ER}_n^{\mathrm{CM}}$ refer to unconditioned ER function for mortality from chronic exposure and age-conditioned ER function for mortality from chronic exposure specific to age cohort n, respectively, and $M^{\mathrm{All}}(M_n^{\mathrm{CPL}})$ and $M^{\mathrm{CPL}}(M_n^{\mathrm{CPL}})$ are mortality rates for all causes for the entire population group (or for age cohort n) and mortality rates for cardiopulmonary diseases for the entire population group (or for age cohort n), respectively. As recommended by Bickel and Friedrich (2005), we assume that chronic mortalities occur only in population groups of age 30 or older. Table 4 displays age-conditioned ER functions used in EPPA-HE.

We compute the number of mortality cases from chronic exposure with the following equation:

$$\mathrm{Cases}_t^{\mathrm{cm}} = \sum_n \mathrm{ER}_n^{\mathrm{CM}} \cdot \left(\sum_{i=a_n}^{t} C_i\right) \cdot M_{nt} \cdot P_{nt} \cdot U_t \tag{4}$$

where a_n, C_t, M_{nt}, P_{nt}, and U_t refer to average birth year for cohort n, PM_{10} concentration level at time t, mortality rate and population size for cohort n and time t, and urbanization ratio at time t, respectively. When valuing mortality from chronic exposure, we use average wage levels for China's employed population, which are endogenously determined within the EPPA–HE model. Both unit labor hour loss and unit leisure time loss are evaluated at the same level as the wage rate, because each worker under the EPPA-HE framework chooses to enjoy his/her leisure time, only at the expense of his/her working time (i.e., lost wage is the opportunity cost of leisure).

[6] Our study considers premature deaths caused by chronic exposure to PM only, and ignores the potential long-term effect of ozone on mortalities. A recent U.S.-based study by Jerrett et al. (2009) found a positive relationship between the long-term exposure to ozone and the risk of death due to respiratory causes, but in the absence of other supportive empirical studies, we conclude that this potential ER relationship still involves high uncertainty, and thus it is premature to apply results from this study to our analysis.

Table 4
Age-conditioned ER Functions for chronic mortalities, China.

Age cohort	30–44	45–59	60–69	70–79	>80
ER functions for chronic mortalities, as % increase in mortality rate	0.089	0.138	0.224	0.295	0.349

Source: Computed from Pope et al. (2002) and Ministry of Health, People's Republic of China (2004).

5. Simulation and results

5.1. Scenarios for EPPA–HE

We use four scenarios to estimate the costs of air pollution. One scenario is the reference case that we call *Historical*. Ozone and PM_{10} concentration levels in this scenario are set at historical levels as described in section 3.3, and the gross domestic product (GDP) numbers that EPPA–HE simulates under this scenario are calibrated to the reported levels for the 1970 to 2005 period. In other words, this reference scenario simulates the reality where observed economic results are already distorted by air pollution effects. The second scenario, which we title *Green*, is a counterfactual case where ozone and PM_{10} concentration levels are set at 20 μg/m^3 and 0.001 μg/m^3, respectively. These concentration levels in the *Green* scenario represent background levels for the two pollutants in the absence of anthropogenic sources of pollutant emissions (Seinfeld and Pandis, 1998). We use the *Green* scenario to capture the full impact of anthropogenic emissions, but the background levels in the scenario would be essentially impossible to attain in reality and thus are not intended as a policy target. The last two scenarios are the *Policy* 1&2 scenarios, which assume modest levels of air quality improvement and thus are more feasible goals of air quality regulations. The *Policy* 1 scenario sets the O_3 level at 70 μg/m^3, which several studies (e.g., Holland et al., 2005) adopt as a cut-off value for health effects, and the PM_{10} concentration level at 40 μg/m^3, which China has adopted as the Class I standard value for residential areas since 1996 (SEPA, 1996). The *Policy* 2 scenario sets the O_3 level at 70 μg/m^3 and the PM_{10} level at 20 μg/m^3, which WHO (2005) recommends as an annual guideline value.

5.2. Simulation results and analysis

Our simulation results show that air pollution has produced substantial socio-economic costs in China. We measure the

Table 5
Estimated costs of anthropogenic air pollution in China: difference in simulation outcomes of *Historical* and *Green* scenarios.

Year	Consumption loss		Welfare loss	
	Billions of 1997 US$	% of historical consumption level	Billions of 1997 US$	% of historical welfare level
1975	15.6	23.1	21.8	14.1
1980	17.4	19.3	23.0	11.2
1985	23.2	15.4	31.1	9.1
1990	23.6	11.0	31.0	6.5
1995	36.7	9.4	52.5	5.9
2000	47.8	8.1	70.8	5.2
2005	69.0	7.3	111.5	5.0

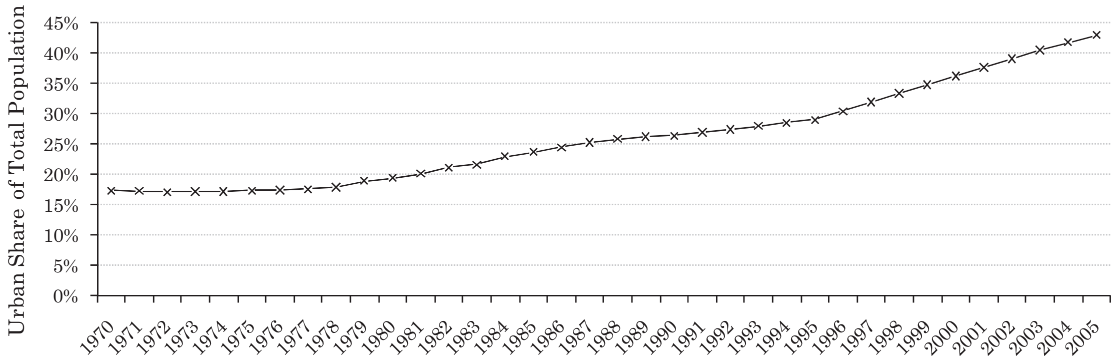

Fig. 7. China's urbanization ratios, 1970–2005.
Source: Data from ACMR (2010).

Table 6
Estimated benefits from hypothetical air quality control measures in China: difference in simulation outcomes of *Historical* and *Policy* 1&2 scenarios.

Year	Policy 1 compared to historical				Policy 2 compared to historical			
	Δ Consumption		Δ Welfare		Δ Consumption		Δ Welfare	
	bn US$[a]	%[b]	bn US$[a]	%[b]	bn US$[a]	%[b]	bn US$[a]	%[b]
1975	11.5	17.1	16.2	10.5	12.5	18.5	17.5	11.3
1980	12.9	14.3	17.2	8.4	13.9	15.5	18.6	9.0
1985	16.4	10.9	22.1	6.5	18.0	12.0	24.2	7.1
1990	14.5	6.7	19.0	4.0	16.6	7.7	21.8	4.6
1995	22.5	5.8	32.9	3.7	26.1	6.7	37.9	4.2
2000	27.0	4.6	41.4	3.1	32.3	5.5	48.9	3.6
2005	38.0	4.1	66.4	3.0	46.6	5.0	78.6	3.6

[a] Billions of 1997 US$.
[b] % to historical consumption (or welfare) level for each year.

pollution health cost in terms of consumption loss, which captures net wages lost due to pollution but does not include leisure time value, and of welfare loss, which is a combination of losses in consumption and leisure time, evaluated as a change in equivalent variation. The consumption loss is also measured an equivalent variation change but excluding the leisure change, and therefore capturing only the market effects.

Table 5 displays the cost of air pollution in China, which is based on the comparison of the simulation outcomes of the *Historical* and *Green* scenarios. This comparison is to estimate the total magnitude of health damage from all kinds of anthropogenic air pollution sources. In terms of consumption, we estimate that for the three decades from 1975 to 2005, air pollution in China reduced annual consumption levels between 7% and 23%. During this period, the consumption loss in absolute terms continuously increased from US$16 billion in 1975 through US$24 billion in 1990 to US$69 billion in 2005.[7]

One explanation for the continued absolute rise of consumption losses is that China experienced rapid urbanization during the period (Fig. 7), and the growth in the exposed urban population, which is affected by PM concentrations, offset the improvement in PM concentrations in the air. In addition, as the productivity of labor increased over the time period, the costs from lost labor were higher for more recent time periods. In relative terms, the consumption loss declined from 23% of the historical consumption level in 1975 to 7% of the historical level in 2005. This is explained by the fact that the overall Chinese economy grew at a much faster rate than the absolute value of the pollution-induced consumption loss. The pollution cost measured as welfare loss shows a similar trend to the consumption loss. Between 1975 and 2005, the calculation of China's lost welfare grew from US$22 billion to US$112 billion annually. In relative terms, this represents a decline from 14% to 5% of the historical welfare levels during this period, for the same reasons outlined for the relative decline in consumption losses.

Table 6 displays net-differences in consumption and welfare levels between the simulation outcomes of the *Historical* and the *Policy* 1&2 scenarios. From this comparison, we derive the benefit the Chinese economy could have enjoyed if it had achieved a set of feasible air quality control targets. In this analysis, we do not consider the costs that would be required to implement the air quality control measures. We estimate that annual benefit from the air quality control targets described in *Policy* 1&2 would be US$12 billion to US$47 billion in terms of consumption increase, or 4% to 19% of the historical consumption levels for the period of 1975–2005. As *Policy* 2 assumes more stringent mitigation of air pollution than *Policy* 1, benefit from the former (US$13 billion to US$47 billion) was larger than that from the latter (US$12 billion to US$38 billion). In addition, when measured in welfare terms, the annual benefit from the two *Policy* scenarios for the same time period is estimated to range between US$16 billion and US$79 billion or between 3% and 11% of the historical welfare levels.

Similar to the trend of the cost of anthropogenic air pollution, displayed in Table 5, benefit in absolute dollar terms increased with time, while benefit in relative terms (% of historical consumption levels) declined with time. More specifically, the absolute gain in consumption (welfare) from the *Policy* 1 scenario grows from US$12 billion (US$16 billion) in 1975 to US$38 billion (US$66 billion) in 2005, meaning a decrease in relative terms from 17% (11%) in 1975 to 4% (3%) in 2005 of the historical levels. Analogously, the consumption (welfare) gain from the *Policy* 2 scenario increases in absolute terms from US$13 billion (US$18 billion) in 1975 to US$47 billion (US$79 billion) in 2005, while it decreases in relative terms from 19% (11%) in 1975 to 5% (4%) in

[7] Unless mentioned otherwise, US$ refers to 1997 US$ throughout this paper.

Table 7
Pollution-induced health outcomes by pollutant, China, 2005 (Unit: thousands of cases).

Health outcomes	O_3	PM_{10}
Respiratory hospital admission	1259	429
Cerebrovascular hospital admission	n/a	307
Cardiovascular hospital admission	n/a	265
Respiratory symptom days	3,322,579	1,913,737
Mortality from acute exposure	166	202
chronic bronchitis	n/a	1,004
Chronic cough (only for children)	n/a	30,024
Cough and wheeze	228,940	3,526,068
Restricted activity day	n/a	2,654,697
Congestive heart failure	n/a	106
Asthma attacks	17,277	n/a
Bronchodilator usage	266,080	255,050
Mortality from chronic exposure (those who died in 2005 only)	n/a	2,742

Table 9
Decomposition of welfare loss in 2005.

	Monetary value (billions of 1997 US$)	Share of total welfare loss (%)
Total welfare loss	69.0	100.0
Direct loss due to mortalities from chronic exposure	42.6	61.7
Mortalities that occurred in 2005	4.4	6.4
Loss in 2005 from prior year cases	38.2	55.4
Direct loss due to other health outcomes	6.7	9.7
Non-fatal health outcomes	6.5	9.4
Mortalities from acute exposure	0.2	0.3
Broader economic losses	19.7	28.6

2005. As explained before, the primary driving factors behind these trends are the interactions among the following three variables: (i) net difference in air quality between the *Historical* and the *Policy* 1&2 scenarios, which substantially declined with time, in the case of PM_{10}, (ii) the size of the Chinese urban population, and (iii) labor productivity growth.

5.3. Decomposition analysis

In this section, we decompose pollution-induced health costs, which we call pollution health costs, by health-end point, pollutant, and cost category. Table 7 displays the number of cases of pollution-induced fatal and non-fatal outcomes in 2005. Then, Table 8 provides the monetary value of the health damage, decomposed by health-end point category, pollutant, and cost category. Pollution health costs displayed in the table are computed by summing up (i) medical expenses spent to recover initial health conditions, (ii) the value of labor time lost due to illness or premature deaths, and (iii) the value of leisure time lost due to illness or premature deaths. We estimate that pollution health costs in 2005, induced by the portion of O_3 and PM_{10} concentrations exceeding their natural levels due to anthropogenic pollution sources, are US$26 billion. Around 87% of the total costs is attributable to excess PM concentrations, and the remaining 13% is from excess ozone concentrations. Leisure loss (52%) and medical expenses (47%) account for most of the ozone-related costs. The medical expenses category (58%) accounts for the largest portion of PM-related costs, and is followed by leisure loss (33%) and wage loss (9%). The morbidity category (82%) is estimated to generate a larger amount of pollution health costs than the mortality category (18%). This result seems to be related to our assumptions in valuation that (i) medical expenses are not involved in the mortality category and (ii) primary victims of fatal damage from acute exposure are those whose remaining life expectancy is at most 6 months even in the absence of exposure to air pollution, which may lead to low valuation of mortality-related pollution costs compared with other studies. Other valuation approaches often use a statistical value of life, and apply it to any mortality in contrast to our approach that uses a years-of-lost-life approach.

Table 9 displays decomposed total air pollution welfare loss in 2005. As mentioned before, welfare includes consumption and leisure. Thus our analysis here does not consider medical expenses, which are redistributed from households to medical service providers within the economy. We split the total welfare loss into three categories: (i) direct loss due to chronic exposure, (ii) direct loss due to other health outcomes, such as mortalities from acute exposure and morbidities, and (iii) broader economic losses, estimated by subtracting the sum of the first two cost categories from the total welfare loss.

The portion of welfare loss from chronic exposure refers to the amount of the consumption and leisure lost by those who died prior to 2005 due to chronic exposure to PM_{10} but would still have survived in 2005 in the absence of air pollution exposure. For this computation, we track past chronic mortalities back from 1959 to 2004; 1959 is the first year when the relevant age cohort for this computation existed, because those who died at age 30 in 1959 would have been 75 in 2004 without excess PM concentrations. Note, again, our assumptions that premature deaths from chronic exposure happen only to the age group of 30 years or older, and leisure time for those who are over 75 years old does not carry any monetary value. We estimate that the direct costs due to mortalities from chronic exposure account for 62% of the total welfare loss, and the direct costs from mortalities from acute exposure or morbidities account for 10%. The remaining portion of the welfare loss (i.e., the broader economic losses category), which is not accounted for by morbidities and mortalities, is US$20 billion or 29% of the total welfare loss. In our EPPA-HE framework, this residual portion is caused by (i) cumulative impacts of welfare loss at one point in time (e.g., lower gross income in one year will lead to less consumption and investment in later years) and (ii) failure to reach the economy's most efficient equilibrium due to pollution-induced distortions in resource allocation (i.e., pollution interrupts

Table 8
Decomposition of pollution health costs, China, 2005 (Unit: millions of 1997 US$).

Health outcome category	Ozone			PM_{10}		
	Medical expenses	Wage loss	Leisure loss	Medical expenses	Wage loss	Leisure loss
Non-fatal health outcomes	1,507	14	1,566	12,881	1,089	3,819
Mortality from acute exposure	n/a	25	85	n/a	31	103
Mortality from chronic exposure (year 2005 only)	n/a	n/a	n/a	n/a	936	3,499
Sub-total	1,507	40	1,650	12,881	2,056	7,421
Sub-total by pollutant		3,197 (13%)			22,358 (87%)	
Total	25,555 (100%)					

Explicit pollution health costs do not include pollution-induced residual cumulative impacts.

Table 10
Static estimates of China's air pollution by the world bank.

Studies	Year of analysis	Estimated costs of air pollution (1997 prices)	Note
World Bank (1997)	1995	US$33.9 billion (4.6% of GDP)	• Considers PM_{10} only and without threshold effects • Based on the Willingness-to-pay valuation method • Omits leisure loss
World Bank and SEPA (2007)	2003	US$54.6 billion (3.8% of GDP)	• Considers PM_{10} only, and adopts 15 $\mu g/m^3$ as lower threshold value for PM_{10} effects • Based on the Willingness-to-pay valuation method • Omits leisure loss

Original estimates are converted to 1997 US$ values by using official foreign exchange rate and consumer price index statistics.

equilibrium because it reduces the amount of labor input and resources available for other production sectors by the portion used by the pollution-health service sector).

Our decomposition analysis also shows that a static estimation method will lead to substantially underestimated estimates of the cost of air pollution. We estimate that a large fraction (84%) of the 2005 total pollution-induced welfare loss in China is from chronic mortalities that happened in the past, cumulative impacts of welfare loss, and distorted resource allocation, which a static analysis often ignores or is unable to capture. Given that the portion of the broader economic losses category for the European region was 12% of the region's total welfare loss in 2005 (Nam et al., 2010), the comparable number for China (29%) suggests that the static analysis for fast-growing economies may further underestimate the cost of pollution, as the same amount of reduction in capital stock, for example, would decrease GDP levels for later years by a larger margin in fast-growing economies than in stagnant economies.

5.4. Comparison with previous studies

Among several studies on air pollution in China, the two World Bank studies (1997, 2007) present cost estimates which can be compared with ours in a parallel fashion. In contrast to our dynamic analytical framework, the World Bank studies adopt a static point-estimation technique, where ER functions and valuation tables are applied without considering the cumulative impact of welfare damage that occurred in the past on future welfare levels. As the World Bank studies consider PM impact only, we keep ozone concentration levels constant at 20 $\mu g/m^3$ for new simulations here and thus eliminate ozone's contribution to our cost estimates. In addition, we use total GDP loss, instead of consumption loss or welfare loss, as a metric of our cost estimates for a parallel comparison.

As shown in Table 10, the World Bank studies estimate that damage to human health from air pollution in China was around 4% to 5% of GDP levels between 1995 and 2003. These numbers are substantially smaller than our estimates from EPPA-HE. As displayed in Table 11, we estimate that damage to human health from air pollution in China was around 6% to 9% of GDP between 1995 and 2005.

Table 11
EPPA-HE-simulated GDP Loss from PM_{10} Concentration in China, 1995–2005 (Unit: billions of 1997 US$, %).

	1995	2000	2005
GDP Loss	63.9	77.0	103.9
% to historical GDP Level	8.7	6.9	5.9

6. Sensitivity analysis

In this section, we carry out sensitivity analysis of our simulation results with regard to two sets of parameters with substantial uncertainty. One is the ER functions and the other is TSP-PM_{10} conversion factors.

6.1. Sensitivity analysis with regard to ER functions

Substantial uncertainty may be involved in the ER functions used in our analysis, because ER relationships may differ by time and place, even when pollutants and health end-points are controlled. In this section, we conduct a sensitivity analysis with regard to ER functions by using upper and lower bound values of ER functions acquired from the 95% confidence interval. We compare simulation results based on upper and lower bound ER values with those based on central ER values, and discuss the robustness of our simulation results.

Our sensitivity analysis shows that our cost estimates in dollar terms are highly sensitive to ER functions, but our general conclusion that pollution health cost in China has been substantial and have declined in relative terms still holds. When lower bound values of ER functions are used for simulation, we have much lower estimates for health damage from air pollution in China. In this case, our cost estimates dropped by more than half, compared to our central estimates displayed in Table 5, to US$5 billion to US$15 billion in terms of consumption loss or to US$7 billion to US$23 billion (Table 12). While absolute costs increased, relative costs decreased. Relative consumption costs declined from 7% of the historical consumption level in 1975 to 2% in 2005, and relative welfare costs declined or from 4% of the historical welfare level in 1975 to 1% in 2005. When the upper bound ER values are used, we

Table 12
Sensitivity analysis 1: lower and upper bound values (95% C.I.) of ER functions (Unit: billions of 1997 US$, % of historical consumption/welfare level).

Year	Consumption loss				Welfare loss			
	Lower-bound		Upper-bound		Lower-bound		Upper-bound	
	bn US$	%	bn US$	%	bn US$	%	bn US$	%
1975	4.7	7.0	31.2	46.2	6.8	4.4	42.2	27.3
1980	5.4	6.0	33.2	36.9	7.4	3.6	42.5	20.7
1985	6.8	4.5	46.0	30.6	9.3	2.7	59.6	17.4
1990	6.5	3.0	48.3	22.5	8.6	1.8	61.1	12.8
1995	9.3	2.4	77.5	19.9	13.1	1.5	105.9	11.8
2000	11.1	1.9	101.6	17.3	15.9	1.2	143.9	10.6
2005	14.8	1.6	148.4	15.8	22.6	1.0	228.6	10.3

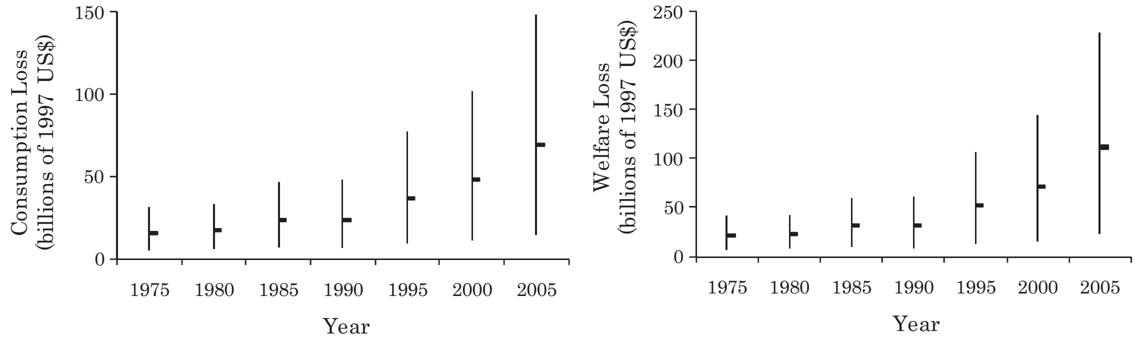

Fig. 8. Error bars of cost estimates produced by uncertainty involved in ER functions: (a) consumption loss, (b) welfare loss.

Table 13
Sensitivity analysis 2: TSP-PM_{10} conversion factor of 65%.

Year	Consumption loss		Welfare loss	
	Billions of 1997 US$	% of historical consumption level	Billions of 1997 US$	% of historical welfare level
1975	15.7	23.2	22.0	14.2
1980	17.5	19.4	23.3	11.3
1985	23.5	15.6	31.7	9.3
1990	24.0	11.2	31.9	6.7
1995	37.8	9.7	55.1	6.1
2000	49.7	8.5	75.5	5.6
2005	73.0	7.8	122.0	5.5

have much higher cost estimates. Our upper bound estimates for absolute consumption (welfare) losses are between US$31 billion (US$42 billion) and US$148 billion (US$229 billion). In sum, our sensitivity analysis of errors in ER estimates suggests error bars of around ±80% (Fig. 8).

6.2. Sensitivity analysis with regard to TSP-PM_{10} conversion factor

Our central estimates for pollution health costs in China, presented in Table 5, are based on the conversion factor between TSP and PM_{10} of 0.5. However, the conversion factor we chose for our central estimates is the most conservative value among those used by other studies. As PM_{10} concentration alone accounts for over 80% of the total pollution health costs, our estimates may be substantially affected by our selection for the conversion factor. In this section, we present simulation outcomes based on the conversion factor of 0.65, which is the highest among those used by other studies.

Table 13 displays simulation outcomes based on the TSP-PM_{10} conversion factor of 0.65. Pollution health costs in China for the period of study were between US$16 billion and US$73 billion, in terms of absolute consumption loss, or between US$22 billion and US$122 billion, in terms of absolute welfare loss. In other words, changing the conversion factor increased the PM_{10} concentration levels by 30%. This, in turn, led to led to 1% to 9% increases in our central cost estimates.

7. Conclusions

Air pollution in China is notorious for its magnitude. In particular, China's PM levels in the 1980s presented a range of 200–317 $\mu g/m^3$, which was at least 10 to 16 times higher than WHO's annual guideline value of 20 $\mu g/m^3$. Even in 2005, when air quality in China was improved substantially, the mean PM_{10} concentration level weighted by the size of urban population was still as high as 109 $\mu g/m^3$. Given that PM is a key air pollutant that accounts for a large fraction of damage to human health, it is not difficult to conclude that air pollution has caused substantial socio-economic burden to China's economy.

In this paper, we apply the method we developed for the US and Europe to China, in order to provide reasonable estimates of socio-economic costs, generated by air pollution in China. Our methodology presents two improvements upon previous work. First, we used a dynamic analysis framework, which allows us to capture certain cumulative dimensions of air pollution's impact on human health. Our method takes into account those aspects which are often ignored by static point-estimate techniques, such as premature deaths that occurred in the past due to chronic exposure to excess PM or the long-term effects of welfare loss at present time on future economic growth. The second aspect of our study that improves on previous work is that we incorporate ozone into our analysis. Ozone is a key pollutant causing substantial health damage, but is excluded from many Chinese pollution studies due to data issues.

Our analysis shows that air pollution in China has created a substantial burden to its economy, though its magnitude in relative terms has gradually declined. We estimate that ozone and PM concentrations beyond background levels have led to US$16 billion to US$69 billion (or 7% to 23%) loss of consumption and US$22 billion to US$112 billion (or 5% to 14%) loss of welfare in China's economy. If China enforced air quality standards described in the *Policy* 1 (or *Policy* 2) scenarios, it would have reaped an increasing benefit, growing from an estimated US$12 billion (US$13 billion) in 1975 to an estimated US$38 billion (US$47 billion) in 2005 in terms of consumption, or US$16 billion (US$18 billion) in 1975 to US$66 billion (US$79 billion) in 2005 in terms of welfare. The sensitivity analysis suggests that our central estimates are highly sensitive to the ER functions, but are robust to our selection of TSP-PM_{10} conversion factor.

8. Discussion

When we compare our study with the two World Bank studies, our estimates present substantially higher values. For example, our estimate of health damage from pollution in 1995 was loss of US$64 billion in GDP (9%), while that of World Bank (1997) was US$34 billion (5%). We end up with higher estimates, primarily because the World Bank studies apply ER functions and valuation tables in a static way, and thus fail to capture the cumulative dimensions of interactions among pollution, human health, and the economy. In fact, our decomposition analysis shows that our estimates for direct economic loss from air pollution, excluding broader economic losses, do not differ much from those in other studies. For example, we estimate that in 2005 China's direct welfare loss from mortalities associated with O_3 and PM exposure was US$42 billion, and World Bank and SEPA (2007) estimate that in 2003 China lost US$41 billion[8] of welfare due to premature deaths associated with PM exposure only.

The comparison of this study with our previous work on Europe shows that economy-wide efficiency loss accounts for a much higher portion of the total pollution-led health damage in China than in Western Europe. Our cost-decomposition analysis shows that 29% of the 2005 pollution-welfare cost for China falls into the broader economic losses category, while 12% was the comparable number for Europe. This suggests that the same amount of current welfare loss from air pollution may have a larger socio-economic impact in the later periods on China than on Europe, because the former is growing at a greater rate than the latter. In other words, applying a static method to fast-growing economies like China may lead to omission of a larger fraction of pollution-induced health damage than applying it to economies at steady state. For this reason, some may find our estimation approach even more valuable for the impact analysis of pollution in emerging economies.

Our study focuses on the benefit side of air quality control, but does not consider the cost that is necessary to achieve such air quality target goals. Future work could conduct a comparably dynamic analysis of the cost dimension, for a complete cost-benefit analysis of air quality control.

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