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Evaluating impacts of air pollution in China on public health: Implications for future air pollution and energy policies

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Abstract

Our objective is to establish the link between energy consumption and technologies, air pollution concentrations, and resulting impacts on public health in eastern China. We use Zaozhuang, a city in eastern China heavily dependent on coal, as a case study to quantify the impacts that air pollution in eastern China had on public health in 2000 and the benefits in improved air quality and health that could be obtained by 2020, relative to business-as-usual (BAU), through the implementation of best available emission control technology (BACT) and advanced coal gasification technologies (ACGT). We use an integrated assessment approach, utilizing state-of-the-science air quality and meteorological models, engineering, epidemiology, and economics, to achieve this objective. We find that total health damages due to year 2000 anthropogenic emissions from Zaozhuang, using the "willingness-to-pay" metric, was equivalent to 10% of Zaozhuang's GDP. If all health damages resulting from coal use were internalized in the market price of coal, the year 2000 price would have more than tripled. With no new air pollution controls implemented between 2000 and 2020 but with projected increases in energy use, we estimate health damages from air pollution exposure to be equivalent to 16% of Zaozhuang's projected 2020 GDP. BACT and ACGT (with only 24% penetration in Zaozhuang and providing 2% of energy needs in three surrounding municipalities) could reduce the potential health damage of air pollution in 2020 to 13% and 8% of projected GDP, respectively. Benefits to public health, of substantial monetary value, can be achieved through the use of BACT; health benefits from the use of ACGT could be even larger. Despite significant uncertainty associated with each element of the integrated assessment approach, we demonstrate that substantial benefits to public health could be achieved in this region of eastern China through the use of additional pollution controls and particularly from the use of advanced coal gasification technology. Without such controls, the impacts of air pollution on public health, presently considerable, will increase substantially by 2020.

Keywords: Air pollution impacts; Public health; China; Energy policy

1. Introduction

Air pollution has become one of the most visible environmental problems in China due to massive coal combustion with inadequate emission controls. An understanding of the link between energy

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consumption and technologies, air pollution and related environmental impacts is necessary to evaluate different air pollution control options but is lacking in China's current policy decision making. Our objective is to establish such a link by quantifying the impacts of air pollution in eastern China on public health in 2000, and the benefits in improved air quality and health that could be obtained by 2020, relative to business-as-usual (BAU), through the implementation of best available end-of-pipe environmental controls (BACT) and advanced coal gasification technologies (ACGT). This comparative health benefit assessment provides an important input to the energy and environmental policy-making process necessary to maximize benefits of regulatory actions or polices. It should be of interest to energy and environmental authorities and local governments in charge of energy and environmental planning in China.

We use an integrated assessment approach which utilizes state-of-the-science air quality and meteorological models, engineering, epidemiology, and economics. A similar approach has been used in other studies examining the environmental impacts and/or costs associated with energy use (e.g., Aunan et al., 2000, 2004; Delucchi, 2000; EPA, 1997, 1999; Feng, 1999; Kunzli et al., 2000; Levy et al., 1999; Li et al., 2004; Lvovsky et al., 2000; Ogden et al., 2004; Rabl and Spadaro, 2000; Rowe et al., 1995a, b; Wang, 1997). However, these earlier studies either focus on specific energy end-use sectors (e.g., coal-fired power plants or transportation) or fuel types (e.g., coal and biomass fuel development), or a policy program such as the Clean Air Act in the United States. Our study makes some major advances in this approach which are highlighted here. First, we have developed an emission inventory with high spatial and temporal resolution that includes both sector specific anthropogenic and biogenic emissions for 2000 and three emission scenarios for 2020 [see (Wang et al., 2005) for details]. Second, we use a multi-pollutant, multi-scale air quality model, the Community Multi-scale Air Quality Modeling System (CMAQ) Version 4.3, to simulate ambient concentrations of pollutants across a multi-province domain. CMAQ simulates atmospheric and land processes that affect the transport, transformation, and deposition of atmospheric pollutants (Byun and Ching, 1999) and explicitly accounts for the formation of secondary particulate matter (PM) which has a significant impact on public health. Third, we use concentration-response (CR) functions from long-term air pollution exposure studies for our health impact assessment. The long term air pollution exposure studies consistently show that the health effects from chronic exposure are nearly an order of magnitude higher than those due to acute exposure alone (Abbey et al., 1999; Dockery et al., 1993; Hoek et al., 2002; Pope III et al., 2002). Fourth, we measure premature mortality based on both the number of deaths and on the years of life lost (YOLL) due to air pollution exposure because differing views exist on the validity of both metrics (e.g., EPA, 1999; Rabl, 2003). When an individual dies prematurely due to long term exposure to air pollution, he or she may lose only a few years of his or her life. Thus, depending on whether economic valuation is based on number of lives lost or YOLL, the perceived health benefits of an air pollution control project may vary sufficiently to alter the results of a cost-benefit analysis. However, this paper only includes an economic valuation of premature mortality based on the number of deaths because there is no consensus on a methodology for estimating the economic value of a YOLL. However, a valuation of mortality based on YOLL is shown in Wang (2004).

Our paper is structured as follows. Section 2 describes the methods used to calculate the changes in ambient concentrations, health impacts and associated economic costs. Section 3 presents results of and Section 4 examines uncertainties in the integrated assessment. Section 5 summarizes our main conclusions.

2. Integrated assessment approach

2.1. General framework

Our integrated assessment includes six steps: (1) define the study region and energy technology scenarios, (2) estimate emissions of air pollutants for 2000 and three scenarios for 2020, (3) simulate ambient air pollution concentrations and distributions, (4) estimate human exposure to air pollutants, (5) estimate health impacts and (6) quantify the economic costs of those impacts. The first three components have been described in detail in Wang et al. (2005) and are summarized below. The other components are described in detail here.

2.2. Defining the study region and energy technology scenarios

We select Zaozhuang Municipality in Shandong Province of eastern China as a case study because its coal-dominated energy structure and development level are representative of many cities in China. Zaozhuang has rich coal reserves, and coal accounts for more than 80% of its primary energy consumption. The Zaozhuang population was 3.5 million in 2000 and is expected to increase by 17% in 2020; its per capita gross domestic product (GDP) was \$842 in 2000 and is expected to increase to \$4008 in 2020 (Zheng et al., 2003).

The region over which we quantify the health impacts of air pollution resulting from energy use in Zaozhuang includes and surrounds Zaozhuang (solid green square in Fig. 1). The total population in the model region was 281 million in 2000.

In addition to the base year 2000, three types of energy and environmental control scenarios for 2020 are examined: BAU, which implies the continuation of conventional coal combustion technologies used in 2000 with limited environmental controls, addition of best available emission control technologies (BACT) to the conventional combustion technologies in Zaozhuang, and the substitution of ACGT. These three scenarios are summarized in Table 1. We include ACGT because of its potential future strategic importance to China. ACGT would facilitate continued use of China's enormous carbon and sulfur rich coal reserves while nearly eliminating emissions of air pollutants and permitting underground sequestration of CO_2 (Larson and Ren, 2003; Williams, 2001; Williams and Larson, 2003; Zheng et al., 2003). All technology scenarios we consider are centered on coal and are designed to meet the same level of energy service demand and socio-economic development projected by the local governments. Energy service demand in 2020 is projected to increase by 150% over 2000 (Zheng et al., 2003). When replacing BAU technologies in Zaozhuang in 2020, BACT are assumed to cover all sectors; ACGT are projected to penetrate 24% of the energy service market in Zaozhuang and provide 2% of the energy needs in three surrounding municipalities with the rest of energy service demand in the modeling domain still met with BAU technologies (Wang, 2004; Wang et al., 2005; Zheng et al., 2003). Our results would need to be adjusted if actual ACGT penetration rates are larger or smaller.

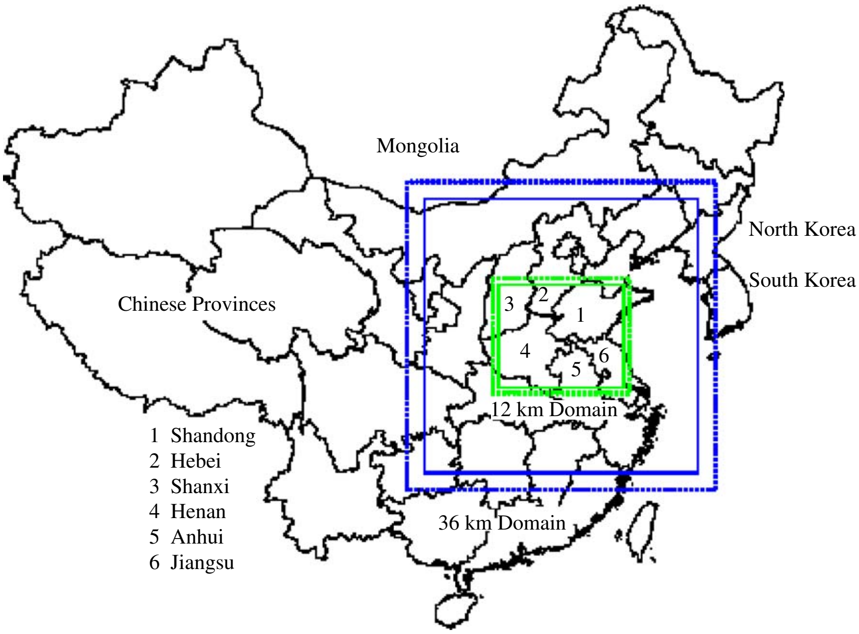

Fig. 1. Map of China and model boundaries. Note: The solid green rectangle demarcates the CMAQ domain with an area of $792 \times 648\,km^2$ and a grid size of $12 \times 12\,km^2$ on which the health impact analysis is focused; the solid blue rectangle demarcates the CMAQ domain with an area of $1728 \times 1728\,km^2$ and a grid size of $36 \times 36\,km^2$ used to provide boundary conditions for the inner region. The dashed green and blue rectangles represent the MM5 model domains. Provinces labeled with numbers are those for which a high-resolution emission inventory has been compiled (Wang et al., 2005).

Table 1
Summary of 2020 energy technology scenarios (see Wang et al., 2005 for details)

Scenario	Main characteristics
BAU	Energy and environmental control technologies and emission factors maintained at the year 2000 level.
BACT	Energy technologies same as year 2000, but equipped with best available end-of-pipe controls such as desulphurization for power plants and catalytic converters on vehicles, specifically: • Power generation: continue to use low-sulfur coal (0.8% S content) as in BAU, SO_2 emissions cut by 90% (Zheng et al., 2003) and emissions of all other species by 20% (estimated by Wang et al. (2005) in coal-fired power plants • Transport sector: CO, NO_x and VOC emissions cut by 75% (Zheng et al., 2003), emission factors for other pollutants same as in 2000 • Residential and industrial sector: emissions of all species cut by 20% (estimated by Wang et al., 2005)
ACGT	Replace conventional coal combustion technologies with advanced coal gasification technologies with 24% penetration in Zaozhuang which supplies 2% of total energy needs (10% of the energy needs in the residential and commercial sectors) in three surrounding municipalities—Jining and Linyi in Shandong Province and Xuzhou in Jiangsu Province. The market share of the ACGT products are described in Wang et al. (2005). Syngas, an intermediate energy product from coal gasification is burned for heat in the industrial sector, and used to generate electricity and produce dimethyl ether (DME) as residential fuel and DME and methanol as transport fuels. • Power generation: although more abundant high-sulfur coal (3.7% S content) is used, SO_2 and other emissions are cut by approximately 99% from affected power plants in Zaozhuang (Zheng et al., 2003). • Transport sector: CO, NO_x and NMVOC emissions from methanol are 80% less than gasoline, and from DME 92% less than diesel (Zheng et al., 2003). • Residential and industrial sector: SO_2, CO and PM emissions from DME are nearly zero.

Final energy demand in 2020 is the same for all scenarios and is described in (Wang et al., 2005).

A high resolution emission inventory was developed for the study region (Wang et al., 2005). The emission inventory includes annual total emissions at the municipality level of carbon monoxide (CO), ammonia (NH_3), nitrogen oxides ($NO_x = NO + NO_2$), NMVOC (non-methane volatile organic compounds), sulfur dioxide (SO_2), and particulate matter smaller than 2.5 μm ($PM_{2.5}$) and smaller than 10 μm (PM_{10}). The Sparse Matrix Operator Kernel Emissions Modeling System (SMOKE) Version 1.3 was used to create the spatial and temporal distribution and chemical speciation of the emission inventory that was used in CMAQ for this analysis. Wang et al. (2005) concludes that emissions of NH_3 are projected to be 20% higher, NMVOC 50% higher, and all other species 130–250% higher in 2020 BAU than in 2000. Both alternative 2020 emission scenarios would reduce emissions relative to BAU. Adoption of ACGT which meets only 24% of energy service demand in Zaozhuang and provides 2% of energy needs in three surrounding municipalities in 2020 would reduce emissions more than BACT with 100% penetration in Zaozhuang.

2.3. Simulating ambient concentrations

CMAQ takes emissions and meteorology as input and simulates hourly ambient concentrations of more than 70 chemical species. Meteorology is generated using the fifth-generation NCAR/Penn State Mesoscale Model (MM5) Version 3.5. A detailed description of the MM5 and CMAQ configurations is provided in Wang et al. (2005).

Changes in annual ambient concentrations required to evaluate the health impacts of air pollution are calculated as the difference between two CMAQ simulations. First, in order to represent each season, we conduct CMAQ simulations for 3–18 of January, April, July and October 2000 and 2020 using the same meteorology for both years. The first 4 days of each month are used as model spin-up and are discarded. We average concentrations in the surface layer (18 m thick) over the four months to obtain annual average pollutant concentrations necessary to evaluate the health impacts due to pollution exposure.

Wang et al. (2005) finds that total $PM_{2.5}$ concentrations are highest in January and lowest in July as a result of higher emissions of $PM_{2.5}$ and its precursors such as SO_2 and NO_x in January. High $PM_{2.5}$ concentrations occur in areas where emissions are large due to high population density and/or industry. The 2020 BAU $PM_{2.5}$ concentrations are projected to be much higher than concentrations in 2000 in all four seasons.

Wang et al. (2005) also evaluated the simulated PM2.5 and SO_2 concentrations for 2000 by comparing them with available observations. The simulated concentrations agree reasonably well with observations in October, but the model frequently under-predicts surface concentrations in April, and, to a lesser extent, in July. The underestimates could potentially be due to several factors, including a mismatch of geographical coverage of the model and the observations, missing sources in our emission inventory including an omission of desert dust, and/or a lack of specific Chinese emission characteristics for some pollution sources.

2.4. Estimating population exposure

Due to long-range transport, the emission of air pollutants in Zaozhuang affects populations residing both inside and outside the city. We have therefore defined a study region which includes and surrounds Zaozhuang (Fig. 1). Within each grid box of our domain we calculate total exposure using both the population and the change in pollutant concentrations occurring between two simulations. Our analysis may slightly underestimate total impacts by excluding people exposed to air pollution originating from Zaozhuang but residing outside of the model region.

The 2000 population is collected by county (The University of Michigan China Data Center, 2003) and assigned to grid boxes (12×12 km each) within the model region using an area weighting factor. The population in Shandong Province is predicted to increase by approximately 17% between 2000 and 2020 (Zheng et al., 2003). We apply this growth rate to the population in each grid box of our domain.

Epidemiological studies from which we obtain CR functions often target specific age groups of a population. We include the same age groups for individual health endpoints as in the original studies. We use the age distribution of the national Chinese population (China Statistics Administration, 2002) to represent the age distribution within each province. For total mortality due to $PM_{2.5}$ exposure, only those age 30 and above (53% of the total population) and infants are included in the analysis. This does not imply that air pollution has no effect on those aged 1–29 years; rather, they are excluded from our analysis because CR functions are not available. However, excluding the population aged 1–29 results in only a small underestimate of the effects of air pollution exposure because the age-specific mortality for this age group is very low and the relative risk from Pope III et al. (2002) seems to be independent of age (Krewski et al., 2000).

2.5. Estimating total health impacts

We include both mortality and morbidity effects. Death and YOLL are both included as measures of mortality; illness is the measure of morbidity. We select PM ($PM_{2.5}$ or PM_{10}) as a surrogate pollutant for estimating overall health impacts because it is believed that PM is responsible for the largest attributable fraction of mortalities due to air pollution exposure and because eastern China suffers from particularly elevated PM levels. We recognize that different components of PM may result in differing health impacts (Hurley et al., 2005), however, the current literature is not sufficient to permit us to characterize these impacts. There is no need to include other pollutants such as SO_2, NO_2, or CO as the concentrations of these pollutants are often correlated with PM and inclusion of the impacts of all pollutants individually would potentially overestimate the contribution of air pollution to mortality and morbidity (Kunzli, 2002). In addition, we calculated the acute effect of O_3 exposure using time-series concentration–response relationships and found the effect to be negligible for our modeling scenarios (Wang, 2004).

Given that in the observed range of ambient concentrations, the relationship between concentrations and health outcomes is approximately linear without a threshold below which no adverse health effects are expected (Daniels et al., 2000; Dominici et al., 2003; Pope III, 2000b; Samoli et al., 2003), total mortality and morbidity due to air pollution exposure is calculated as follows:

$$\Delta\text{cases} = I_{\text{ref}}\,\text{POP}\,\gamma\,\Delta C, \tag{1}$$

where I_{ref} is the annual baseline mortality or morbidity rate of the study population, POP the exposed population, γ the CR coefficient, ΔC the changes in annual ambient concentrations due to changes in emissions of air pollutants, and Δcases the additional cases of mortality or morbidity per year due to change in ambient concentration. The CR coefficient (γ) we use for years-of-life-lost (YOLL) already incorporates the Chinese baseline mortality rate (Leksell and Rabl, 2001). Thus the

Table 2
Baseline mortality and morbidity incidence rates in 2000

Health endpoint	Rate[a]	Reference[b]
Total mortality	0.00645	China Statistics Administration (2002)
Mortality among 30+ yr old	0.01013	China Statistics Administration (2002)
Infant mortality	0.0247	China Statistics Administration (2002)
Chronic bronchitis	0.0139	Chen et al. (2002)
Respiratory hospital admissions	0.0124	Chen et al. (2002)
Cardiovascular hospital admissions (>65 yr old)	0.085	Chen et al. (2002)
Acute bronchitis	0.39	Chen et al. (2002)
Asthma attack (< 15 yr old)	0.0693	Chen et al. (2002)
Asthma attack (⩾15 yr old)	0.0561	Chen et al. (2002)
Restricted activity days	19	Ostro (1987)

[a]Units are cases per year per person in the population or for a particular age group as specified.
[b]All rates are for China except restricted activity days.

equation for estimating total YOLL becomes:

$$\Delta \text{cases(YOLL)} = \text{POP}\,\gamma\,\Delta C. \quad (2)$$

2.5.1. Baseline mortality and morbidity rates (I_{ref})

We use the national average mortality rate of 0.645% in 2000 (China Statistics Administration, 2002) for the baseline mortality rate for our study region because the municipality-level mortality rates were not available to us. Baseline rates for various morbidity endpoints were neither available at the national level nor for our study region; hence we use baseline morbidity rates for Shanghai which is nearby (Table 2). For restricted activity days no baseline rates are available for China; we therefore use baseline rates from the original studies. The use of baseline mortality rates that are not specific to our region introduces presently unavoidable uncertainty into our calculations.

2.5.2. Concentration-response (CR) coefficients for death and illness

Concentration-response (CR) coefficients for both the premature mortality and morbidity endpoints we use in our analysis are shown in Table 3. Asia differs from the United States and Europe in

Table 3
Concentration—response (CR) coefficients for mortality and morbidity used in this study

Health endpoints	Pollutant ($\mu g\,m^{-3}$)	γ[a] (95% CI[b])	Age group	Reference	Study type
Adult mortality	$PM_{2.5}$	0.58% (0.2–1.04%)	Age 30+	Pope III et al. (2002)	Cohort
Infant mortality	PM_{10}	0.39% (0.2–0.68%)	27 days to 1 year old	Woodruff et al. (1997)	Cohort
Chronic bronchitis	PM_{10}	0.45% (0.13–0.77%)	All ages	Jin et al. (2000), Ma and Hong (1992)	Time-series
Acute bronchitis	PM_{10}	0.55% (0.19–0.91%)	All ages	Jin et al. (2000)	Cross-sectional
Cardiovascular HA[b]	PM_{10}	0.1% (0.067–0.15%)	Age 65+	Samet et al. (2000)	Time-series
Respiratory HA[b]	PM_{10}	0.036% (0.012–0.06%)	All ages	Spix et al. (1998)[c]	Time-series
Restricted activity days	PM_{10}	1.5% (0.76–2.35%)	Age 18–65	Cifuentes et al. (2001), Ostro (1990)[d]	Time-series
Asthma attack	PM_{10}	0.39% (0.19–0.59%)	Adults (⩾15 yr)	Chen et al. (2002), Kan and Chen (2004)	Time-series
Asthma attack	PM_{10}	0.44% (0.27–0.62%)	Children (< 15 yr)	Chen et al. (2002), Kan and Chen (2004)	Time-series

[a]Units are % change in mortality and morbidity as a result of a 1 $\mu g\,m^{-3}$ change in PM concentration.
[b]CI = confidence interval; HA = hospital admissions.
[c]Originally based on black smoke (BS), converted to PM10 by multiplying by 0.6.
[d]The baseline morbidity rate has been incorporated into the CR coefficients by Cifuentes et al. (2001).

air pollution composition, the conditions and magnitude of exposure to that pollution, and the health status of exposed populations. However, a recent literature review of time-series studies conducted in Asia found that short-term exposure to air pollution in the studied regions is associated with increases in daily mortality and morbidity effects that are similar to those found in Western countries (HEI, April 2004) supporting transferability of relative risks. In a meta-analysis of short-term mortality studies in China, however, Aunan and Pan (2004) found a lower response to elevated pollution levels than would be predicted by Western country studies. In the absence of long-term cohort studies in high pollution areas, however, they suggest that estimates from US studies may be used in China with the recognition that the results are likely to be on the high side. We choose to use CR coefficients for adult mortality obtained from a cohort study conducted in the United States (Pope III et al., 2002) because no long-term studies have been conducted in China or other developing countries. The outcomes of cohort studies are a combination of acute and chronic effects which are not separable because the outcomes accumulate over long time periods and could be triggered by either cumulative or short-term peak exposures (Dominici et al., 2003; Kunzli et al., 2001). Therefore, cohort studies more accurately represent the full effects of air pollution than do time-series studies. In addition, among the existing cohort studies, Pope III et al. (2002) includes the largest cohort size and area coverage. We include the CR coefficient of PM_{10} mortality for infants (one month to one year old) from Woodruff et al. (1997) which is the only cohort study that examines the association of infant mortality and long-term air pollution exposure.

Studies on the association of morbidity and air pollution exposure are much less comprehensive than mortality. Among the existing morbidity studies, fewer examine chronic morbidity than acute morbidity. As a result, for most morbidity effects, we rely on existing time–series studies (see Table 3), which likely leads to an underestimate of total morbidity. We include morbidity endpoints from Chinese studies or pooled estimates whenever available. The values we use are similar to those reported by Aunan and Pan (2004) though they report higher (lower) values for respiratory (cardiovascular) hospital admissions.

2.5.3. Concentration-response coefficients for years of life lost

Existing epidemiological studies examine the increase of relative risk of premature mortality as a result of exposure to air pollution for a given population, but do not provide the age structure of the premature deaths. Thus, the derivation of YOLL requires assumptions and indirect estimates, and needs to take into account the age distribution, baseline mortality rate, magnitude of change in PM concentrations, relative risk due to changes in PM, and the length of exposure.

Several studies have attempted to estimate the YOLL in mortalities resulting from chronic exposures based on either an actual life table of a population or a demographic model simulating a life table. Essentially, these studies apply the CR coefficient from Pope III et al. (2002) to each age group of a population, calculate the life years lost for each age group given the life expectancy of the population, and then derive the average life years lost for the population. These studies show that for a $10\,\mu g\,m^{-3}$ increase in $PM_{2.5}$ concentration, the YOLL per person exposed for a population age 30 and above is in the range of several months to more than one year (Brunekreef, 1997; EPA, 1997; Leksell and Rabl, 2001; Pope III, 2000a). Since our analysis uses a single year of emission perturbations from different energy technology scenarios to calculate health impacts, we use the results from Leksell and Rabl (2001) for China. Note that the China coefficient shown in Leksell and Rabl (2001) is for age 35 and above. However, using the same coefficient for the study population age 30 and above is assumed to introduce negligible error. For exposure to $1\,\mu g\,m^{-3}$ increase in $PM_{2.5}$ the concentration-YOLL coefficient is 4.7e-4 YOLL for Chinese age 30 and above and 1.66e-5 YOLL for infants 27 days to 1 year old (Rabl, 2003) (based on the CR coefficient of 0.39% from Woodruff et al. (1997).

2.6. Economic costs of premature mortality and morbidity

We estimate the economic costs of premature mortality and morbidity as the product of the number of cases and value per case using the "willingness-to-pay" metric. Willingness-to-pay (WTP) indicates the amount an individual is willing to pay to acquire (or avoid) some good or service. WTP can be measured through revealed preference or stated preference methods. Revealed preference data is either observed or reported actual behavior, and stated preference data is observed or expressed in response to hypothetical scenarios. A commonly used form of stated preference in WTP studies is contingent valuation. Wang et al. (2001) was the only contingent valuation study on the value of a statistical life (VSL) conducted in mainland China that was available at the time of this study. It found that the median WTP value to save one statistical life was $34,583 (1998 US$) in Chongqing City, China in 1998. For comparison, the mean value in the US was $4.8 million (1990 US$) (EPA, 1997). If we only account for the difference in per capita income in 2000 between the US ($34,260) and China ($840) (World Bank, 2001) and assume the VSL is proportional to income, the Chinese VSL in 2000 would be $0.15 million (2000 US$).

We, however, make the conservative assumption that the VSL for Chongqing is representative of China. Given that the inflation rate in China between 1998 and 2000 was −1% and that the per capita income in China is projected to increase from $840 in 2000 to $4008 in 2020, the resulting VSL is $34,235 in 2000 and $163,351 (2000 US$) in 2020. There are great uncertainties involved in VSL valuations which we discuss in Section 4.

There have been very few studies of the WTP to avoid morbidity in China. As a result, we extrapolate from US values, based on the income difference between the two countries. These inferred values may be higher than in-country survey values as in the case of VSL ($0.15 million vs. $34,235). Using the in-country survey value for VSL and the inferred value for morbidity may overweight the importance of morbidity in our results. We thus mechanistically adjust the inferred values for morbidity to be consistent with the in-country VSL by multiplying the morbidity values by the ratio of the in-country VSL ($34,235) to the inferred VSL ($0.15 million). The results are shown in Table 4.

Table 4
Valuation of morbidity for China (2000 US$)

Health endpoints	US values (EPA, 1997)	Chinese values	
		2000	2020
Chronic bronchitis	338,000	1854	8848
Respiratory hospital admissions	8970	49	235
Cardiovascular hospital admissions	12,350	68	323
Acute bronchitis	59	0.5	2
Asthma attack	42	0.2	2
Restricted activity days	108	0.6	3

3. Results and discussion

Emission scenarios used to quantify changes in ambient concentrations of PM in 2020 resulting from the use of different energy technologies are shown in Table 5. Scenarios with zero emissions from Zaozhuang in 2000 (B) and 2020 (D), although unrealistic, are created to quantify the total effect of Zaozhuang's emissions on ambient concentrations across the modeling domain. Scenario B minus A and D minus C provide concentration distributions resulting from anthropogenic emissions in Zaozhuang in 2000 and under 2020 BAU, respectively. Scenario E minus C gives the reduction in emissions resulting from replacing 2020 BAU technologies with BACT in Zaozhuang; scenario F minus C provides the reduction in emissions resulting from

Table 5
Emission scenarios for 2000 and 2020 used in CMAQ simulations

Emission scenario	Year	Technology scenario (market share[a])		
		Zaozhuang	Jining, Linyi and Xuzhou	Rest of the model region
A	2000	BAU (100%)	BAU (100%)	BAU (100%)
B	2000	Zero anthropogenic emissions	BAU (100%)	BAU (100%)
C	2020	BAU (100%)	BAU (100%)	BAU (100%)
D	2020	Zero anthropogenic emissions	BAU (100%)	BAU (100%)
E	2020	BACT (100%)	BAU (100%)	BAU (100%)
F	2020	ACGT (24%) and BAU (76%)	ACGT (2%) and BAU (98%)	BAU (100%)

[a]Share of technology-specific energy products in the final energy market.

replacing 24% of 2020 BAU technologies with ACGT in Zaozhuang.

3.1. Health impacts of Zaozhuang's air pollutant emissions in 2000

Emissions from Zaozhuang not only affect ambient pollutant concentrations in Zaozhuang, but also areas outside of Zaozhuang due to air pollution transport. After hypothetically eliminating all anthropogenic emissions from Zaozhuang, the entire model region experiences a decrease in PM concentrations, and the Zaozhuang source region experiences the largest reduction in both total and secondary $PM_{2.5}$ concentrations (about 10–15 and 2–3 $\mu g\,m^{-3}$ annual average decrease, respectively).

As shown in Table 6, the 2000 anthropogenic emissions of air pollutants from Zaozhuang are estimated to have caused approximately an additional 6000 deaths (5244 adults and 612 infants) in the model region due to total PM exposure, amounting to about 42,000 YOLL. Our simulation indicates that 25% of all deaths resulting from total PM exposure occur in Zaozhuang, equivalent to a 6% increase of its natural mortality rate. Secondary PM is estimated to be responsible for 48% of excess deaths due to PM exposure. This is because secondary PM has a relatively long lifetime and is transported further than primary PM from the source region thus affecting the health of more people outside the source region than does primary PM.

Total health costs are the sum of the economic values of death and illness. The total economic damages of the resulting health impacts from 2000 are estimated to be US$0.28 billion. This is equivalent to 10% of Zaozhuang's 2000 GDP. The economic damage of illness accounts for 29% of the total health damages.

Health damages caused by coal use can be compared with the market price of coal. The current coal price does not include the external cost to health and the environment. Zaozhuang consumed 3.1 million tons of coal in 2000 and coal accounted for 82% of its total energy consumption (Zheng, 2003). We estimate the upper bound of the range of health damages associated with one ton of combusted coal by assuming the emissions from the use of fuels other than coal is negligible; thus the value of damage from coal is equal to the total health damage costs from air pollution divided by the total tonnage of coal consumed. The lower bound of the range is obtained by assuming that the emissions from the use of fuels other than coal is the same as the emissions from coal; thus the health damage

Table 6
Regional health impacts from, and economic costs of, 2000 and 2020 BAU anthropogenic emissions from Zaozhuang and potential health benefits from technology substitution in Zaozhuang in 2020

Year	2000		2020 BAU		*E (BACT)—C (BAU)*		*F (ACGT and BAU)—C (BAU)*	
Pollutant	Total	Secondary PM[a]	Total	Secondary PM	Total	Secondary PM	Total	Secondary PM
Health impacts (100 cases) (% in Zaozhuang)								
Death	−59	−28	−107	−26	−25 (29%)	−8 (8%)	−52 (36%)	−2 (13%)
Years of life lost	−421	−219	−745	−200	−174 (27%)	−63 (8%)	−347 (35%)	−18 (13%)
Chronic bronchitis	−361	−56	−856	−52	−177 (44%)	−16 (8%)	−496 (44%)	−5 (13%)
Acute bronchitis	−1,2390	−1934	−29342	−1771	−6069 (44%)	−553 (8%)	−16994 (44%)	−157 (13%)
Hospital admission	−1569	−245	−3717	−224	−769 (44%)	−70 (8%)	−2153 (44%)	−20 (13%)
Restricted activity day	−59611	−9307	−141170	−8520	−29201 (44%)	−2662 (8%)	−81759 (44%)	−756 (13%)
Asthma attack	−1378	−215	−3263	1197	−675 (44%)	−62 (8%)	−1890 (44%)	−17 (13%)
Health damage costs (million 2000 US$)								
Death[b]	−200	−97	−1750	−423	−404	−132	−841	−38
Illness [b]	−80	−13	−904	−55	−187	−17	−523	−5
Health damage costs as % of Zaozhuang's GDP								
Death and illness	−10%	−4%	−16%	−3%	−4%	−1%	−8%	0

Emission scenarios are defined in Table 5. Negative values for health impacts indicate health damages.
[a]Secondary PM includes sulfates, nitrates, ammonium, and secondary organic carbon, all of which are categorized as $PM_{2.5}$.
[b]Death and years of life lost are two measures of mortality. Illness is the measure of morbidity.

from coal is equal to 82% of the upper estimate. Because coal is the dominant fuel in Zaozhuang, the range derived from this simple approach is narrow and thus provides a meaningful indication of the health damages resulting from coal use. We estimate that each ton of coal combusted in Zaozhuang incurred \$90–\$110 of health damages in 2000. These health damage costs are in striking contrast to the market price of coal in China which was \$30 ton^{-1} (based on a market price of 248 Yuan ton^{-1} with \$1 = 8.3 Yuan) in 1997 (Fridley, 2001). If environmental externalities were reflected in the market price, coal prices in China would have more than tripled.

3.2. Health impacts of Zaozhuang's air pollutant emissions in 2020 BAU

The 2020 BAU anthropogenic emissions from Zaozhuang are estimated to cause approximately 11,000 premature deaths or 75,000 YOLL due to PM exposure in the model region, nearly doubling the 2000 figures (Table 6). Twenty-four percent of the total mortalities resulting from PM exposure are due to secondary PM. Secondary PM is projected to be a smaller fraction of total PM concentrations under 2020BAU than in 2000 due to a projected relative increase in primary PM emissions in 2020 due to increases in residential coal use (Wang et al., 2005). As a result, the percentage of mortalities attributed to secondary PM under 2020 BAU is lower than that in 2000. Zaozhuang, the emission source, is estimated to bear 33% of the total premature mortalities resulting from PM exposure causing a 13% increase in baseline mortality rates. Sixteen percent of the excess deaths result from secondary PM exposure (compared with 48% in 2000).

The total economic value of the health damages resulting from the 2020 BAU anthropogenic emissions from Zaozhuang are estimated to be \$2.7 billion. This is equivalent to 16% of the projected 2020 GDP in Zaozhuang and is 10 times larger than the 2000 value due to projected increases in energy consumption and values of mortality and morbidity.

We estimate the health damages associated with one ton of coal combusted using the same approach for 2020 BAU as for 2000. Zaozhuang is projected to consume 11.5 million tons of coal in 2020 BAU (Zheng, 2003). We estimate that under 2020 BAU each ton of coal burned in Zaozhuang will incur \$230–\$280 of health damages resulting from air pollution exposure. As the price of coal in the US is projected to be approximately constant from now to 2025 (EIA, 2004), we assume the price of coal in China will also be the same in 2020 as in 2000, approximately \$30 ton^{-1}. If environmental externalities were truly reflected in the market price of coal, in 2020 the price of coal in China should be more than eight times higher than in 2000.

3.3. Health benefits of potential technology changes in Zaozhuang in 2020

Significant benefits, including reduction in emissions, ambient PM concentrations and air pollution exposure related mortalities and morbidities, could be achieved through technology upgrades in Zaozhuang in 2020 (Fig. 2 and Table 6). The benefits from partially switching from BAU to ACGT (F minus C) are much larger than from switching from BAU to BACT (E minus C) except for secondary $PM_{2.5}$ concentrations (Fig. 2d). Higher secondary $PM_{2.5}$ concentrations occur under ACGT than BACT because when dimethyl ether (DME), a product of ACGT, is used to replace coal in the rural residential sector, more NO_x is emitted than under BACT (Zheng et al., 2003). As a result, under ACGT NO_x emissions from the rural areas of Zaozhuang (where the residential sector is a large contributor to total NO_x emissions) are higher than under BACT and result in additional secondary $PM_{2.5}$ formation, even though total NO_x emissions from Zaozhuang under ACGT are lower than under BACT.

The total economic benefit of reduced health impacts resulting from a substitution of E (BACT) for C (BAU) in Zaozhuang are estimated to be \$0.6 billion, nearly half of which would occur in Zaozhuang. The total economic benefit of reduced health impacts resulting from F (ACGT) substituting for C (BAU) in Zaozhuang are estimated to be \$1.4 billion, 60% of which would occur in Zaozhuang itself. These results indicate that about one-fifth to one-half of the total health damages related to air pollution from Zaozhuang in 2020 BAU could be avoided by adopting the BACT or ACGT emission scenarios.

3.4. Health impacts by PM constituent and per kg emission of pollutant

We next attribute health impacts to the constituents of secondary PM and calculate damages per

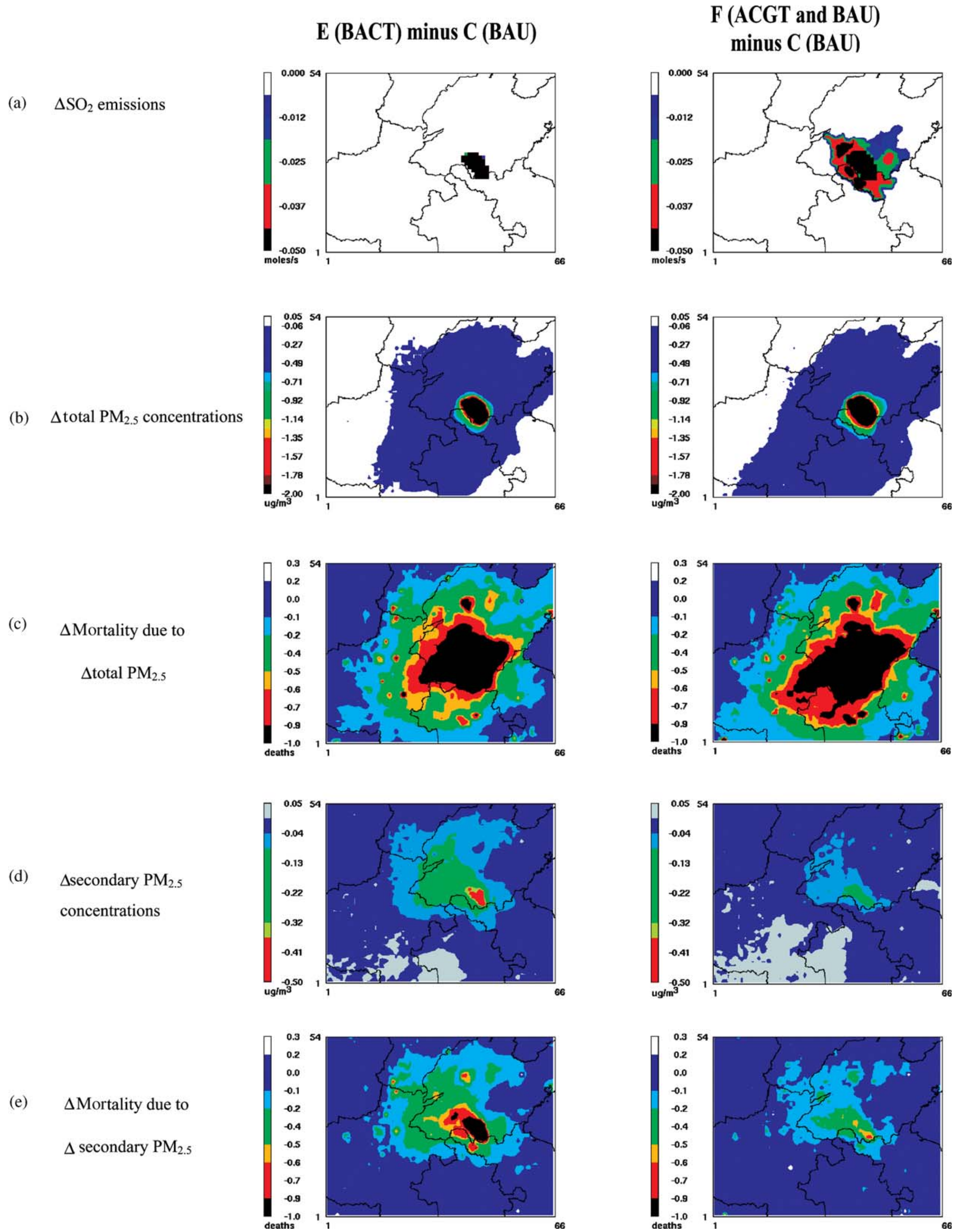

Fig. 2. Effects of implementing BACT and ACGT in Zaozhuang in 2020 on ambient concentrations and resulting mortalities. Although this figure only shows anthropogenic emissions of SO_2, health impacts are calculated following reduction in all anthropogenic emissions from Zaozhuang, including CO, NH_3, SO_2, NO_x, PM_{10}, $PM_{2.5}$ and NMVOC. (a) Δ SO_2 emissions; (b) Δ total $PM_{2.5}$ concentrations; (c) Δ Mortality due to Δ total $PM_{2.5}$; (d) Δ secondary $PM_{2.5}$ concentrations; and (e) Δ mortality due to Δ secondary $PM_{2.5}$.

Table 7
Magnitude of total health impacts and health impacts per kg emitted pollutant attributable to primary PM and key secondary PM constituents (SO_2, NO_x, and NMVOC) for various scenarios

Scenarios compared	Reduction in emissions ($kton\,yr^{-1}$)	Total health impacts (million US$ yr^{-1})	Health impacts[a] (2000 US$ kg^{-1} emissions yr^{-1})
	Primary PM	*Primary PM*	
A to B	220	−158	−0.7
C to D	476	−971	−2.0
C to E	157	−198	−1.3
C to F	536	−589	−1.1
	SO_2	*Sulfates*	
A to B	65	−47	−0.7
C to D	224	−248	−1.1
C to E	115	−149	−1.3
C to F	152	−31	−0.2
	NO_x	*Nitrates*	
A to B	13	−34	−2.7
C to D	99	−106	−1.1
C to E	30	7	0.2
C to F	50	0	0.0
	NMVOC	*Secondary anthropogenic organics*	
A to B	31	0	0.0
C to D	47	0	0.0
C to E	17	0	0.0
C to F	4	0	0.0

Scenarios are defined in Table 5.
[a]Health impacts per kg emissions (2000 US$ kg^{-1} emissions yr^{-1}) = total health impacts (million 2000 US$ yr^{-1})/reduction in emissions ($kton\,yr^{-1}$). Negative values are health benefits; positive values are health damages.

unit of precursor emissions. As shown in Table 7, the health impacts of sulfates in 2000 (2020 BAU) are estimated to be $47 ($248) million due to death and illness. The combined health impacts of sulfates and nitrates dominate the health impacts attributed to secondary PM. The health impacts of secondary anthropogenic organic aerosols are negligible. In both 2000 and 2020 BAU, the health impacts per kilogram primary PM emissions are highest, SO_2 and NO_x emissions second, and NMVOC emissions lowest ("A to B" and "C to D").

In terms of marginal health benefits from technology upgrades, under BACT reducing primary PM emissions provides comparable benefit to reducing SO_2 emissions. Under ACGT reductions in primary PM emissions are of significantly larger benefit. Notably, if Zaozhuang moves from BAU to BACT in 2020, SO_2 emissions decrease leading to increases in nitrate concentrations despite a decrease in NO_x emissions. This leads to net health damages rather than benefits (Table 7). This occurs because aerosol formation is NH_3-limited in the region and when SO_2 emissions decrease, more ammonium is available to form ammonium nitrate. In order to reduce the formation efficiency of ammonium nitrate and thus nitrate concentrations, NH_3 emissions, which are virtually all from the agricultural sector (Wang et al., 2005), must be reduced. Under BAU, BACT and ACGT scenarios, NH_3 emissions are the same. Therefore, we cannot calculate the health benefits of NH_3 reduction here.

Our marginal health benefit calculation suggests that if the primary air pollution control objective is to minimize the total health damages of air pollution from energy use, and if NH_3 emissions remain the same, highest priority should be given to reducing primary PM and second highest priority to reducing SO_2 emissions. Of course, as the air pollution composition changes over time, control strategies will need to be adjusted accordingly. For example, in an advanced energy system dominated by ACGT, SO_2 emissions would be reduced to nearly zero. Much more NH_3 would therefore be available to form ammonium nitrate and controlling NH_3 and NO_x emissions would become more important for reducing secondary PM concentrations. In addition, the control strategies are specific to this region and caution should be taken when applying these findings to other locations.

4. Uncertainty analysis

Uncertainties exist at every step of the integrated assessment: emission estimates, calculated ambient concentrations, epidemiological concentration—response relationships, exposure estimates, health impacts and economic valuation. Despite these uncertainties, we believe this analysis can provide valuable, policy-relevant information on the relative benefits of different future emission scenarios and mitigation strategies.

As discussed in Wang et al. (2005), the major source of uncertainty in emissions is the Chinese government's statistics on energy consumption. Other possible sources of uncertainty in emissions include the use of aggregate coal quality data for an entire province and extrapolation of the PM emission factors for the industrial sector from the US Air Pollution (AP)-42 database.

Changes in ambient concentrations required for our health impact analysis are obtained as the difference between two selected model simulations for which the emissions from Zaozhuang and, in the ACGT case, from three other municipalities, are the only input variables that differ. Thus the accuracy of the changes in PM concentrations in response to changes in energy technologies largely depend on the accuracy of the emission estimates for Zaozhuang. In addition, due to computational limitations, simulations for only 12-days of each season were conducted which limits the variability in meteorology that is represented.

Some in the health impact assessment community argue that it may not be appropriate to apply a relative risk (RR) for PM estimated for one population to another if the baseline health status of the two differs significantly (i.e., US and China) and background pollution levels differ. They argue that using RR for specific cause of death categories would be preferable to using total mortality as was done in this study. Unfortunately, cause-of-death data was not available to us for our region of eastern China which precluded such an analysis. In addition, our health effect estimates do not cover all areas that are potentially affected by air pollution from Zaozhuang, do not account for health endpoints for which CR coefficients from epidemiological studies or monetary valuation are not available, and only include the age groups for which the CR coefficients for mortality and morbidity are available from the original epidemiological studies. As a result, we potentially underestimate the total health impacts of the air pollution from Zaozhuang.

The health impact estimates are affected by the background prevalence of health outcomes. Due to lack of in-country data for China we have used the incidence rates of some morbidity endpoints from the populations in the original epidemiological studies, conducted in the US or in Shanghai, China, which are likely different from those in the study region. If the baseline incidence rates in the original study regions are lower than what we use in our analysis, we potentially overestimate the health impacts of outdoor air pollution. Exclusion of some age groups in calculating air pollution exposure and use of time series studies for morbidity endpoints may also lead to an underestimate of health impacts.

Substantial uncertainty also exists for the economic valuation of premature mortality and morbidity; the WTP approach is heavily debated in the economic literature. The fact that only one survey study on VSL in China was available makes the Chinese VSL estimate of $34,235 used in this analysis highly uncertain. The uncertainty is further illustrated by the fact that this estimate is only a fraction of a percent of the corresponding US value, but is equivalent to 40 times the per capita GDP in China in 2000. However, we believe only more in-country health valuation studies can solve this puzzle.

We conducted an analysis of the aggregate uncertainty embedded in three key input parameters: ambient concentrations, CR coefficients and VSL, using Monte Carlo simulations (see Wang, 2004 for details). We found that the uncertainty range of the excess mortality related to the year 2000 anthropogenic emissions in Zaozhuang is about 40% of the mean value and that the uncertainty range of the economic values of the associated health impacts is about 1.1 times the mean value. Thus we have much higher confidence in the estimates of physical health impacts than in their economic valuation.

5. Conclusions

We have quantitatively estimated the health impacts and damage costs for year 2000 and projected for year 2020 due to anthropogenic emissions of air pollutants from Zaozhuang, China. The 2000 (2020 BAU) anthropogenic emissions in Zaozhuang are estimated to have caused approximately an additional 6000 (11,000) deaths in the model region related to total PM exposure, amounting to approximately 42,000 (75,000) YOLL. A 25% (33%) of all premature mortalities from PM exposure resulting from Zaozhuang's emissions in 2000 and under 2020 BAU are estimated to occur in Zaozhuang. This results in a 6% (13%) increase in the background death rate in the city.

The health costs due to year 2000 anthropogenic emissions from Zaozhuang are estimated to be US$ 0.28 billion, equivalent to 10% of Zaozhuang's GDP. If these health costs were internalized in the market price of coal, the coal price in 2000 would have more than tripled. In 2020, if no additional controls are implemented and energy consumption increases as projected, the resulting health costs due to anthropogenic emissions from Zaozhuang are estimated to be $2.7 billion, equivalent to 16% of the projected 2020 GDP, 10 times larger than in 2000. Although final energy demand is expected to increase by 150% from 2000 to 2020, because of rising incomes the health damage costs due to anthropogenic emissions from Zaozhuang under BAU are estimated to increase 4 times as fast as energy consumption. To include the environmental externalities in the market price of coal in 2020 under BAU, the price of coal would need to be about eight times the projected price.

If Zaozhuang moves from BAU to BACT (with 100% market share of energy services), the total health benefits would amount to \$0.6 billion, which would be a 20% reduction in the total health damages related to air pollution from Zaozhuang under 2020 BAU. Under BACT health damages would be equivalent to 13% of projected GDP. If Zaozhuang moves from BAU to ACGT (with 24% market share of final energy services demand in Zaozhuang and 2% in three surrounding municpalities) instead of BACT, the health benefits from mortality and morbidity avoided would be \$1.4 billion, half of the total health damages related to air pollution from Zaozhuang under 2020 BAU. Under ACGT health damages would be equivalent to approximately 8% of projected GDP.

Our assessment indicates that the economic costs of the health impacts from air pollution arising from coal combustion in eastern China was large in 2000 and is potentially enormous in 2020 if no additional controls are implemented. Public health would clearly benefit from BACT and ACGT and hence better air quality. Furthermore, ACGT are even more effective in controlling local air pollution than end-of-pipe controls and provide an opportunity to sequester CO_2 underground. Our marginal health benefit calculation suggests that if the primary air pollution control objective is to minimize the total health damages of air pollution from energy use and if NH_3 emissions remain the same, the model region should focus on reduction in primary PM and SO_2 emissions.

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References

Abbey, D., Nishino, N., Mcdonnell, W., Burchette, R., Knutsen, S., Beeson, W.L., Yang, J.X., 1999. Long-term inhalable particles and other air pollutants related to mortality in nonsmokers. American Journal of Respiratory and Critical Care Medicine 159, 373–382.

Aunan, K., Pan, X.-C., 2004. Exposure-response functions for health effects of ambient air pollution applicable for China—a meta-analysis. Science of the Total Environment 329, 3–16.

Aunan, K., Fang, J., Vennemo, H., Oye, K., Li, H., Seip, H.M., 2000. Co-benefits from CO_2-emission reduction measures in Shanxi, China—their importance in context of CDM, Policy reform and the environment in China, organized by PACE (Profeesional Association for China's Environment), Washington, DC.

Aunan, K., Fang, J., Vennemo, H., Oye, K., Seip, H.M., 2004. Co-benefits of climate policy—lessons learned from a study in Shanxi, China. Energy Policy 32, 567–581.

Brunekreef, B., 1997. Air pollution and life expectancy: is there a relation? Occupational and Environmental Medicine 54, 781–784.

Byun, D.W., Ching, J.K.S., 1999. Science Algorithms of the EPA Models-3 Community Multiscale Air Quality (CMAQ) Modeling System. EPA/600/R-99/030, Atmospheric Modeling Division, National Exposure Research Laboratory, US Environmental Protection Agency, Research Triangle Park, NC.

Chen, C., Chen, D., Green, C., Wu, C., 2002. Benefits of expanded use of natural gas for pollutant reduction and health improvement in Shanghai. The Sinosphere Journal 5 (2), 58–64.

China Statistics Administration, 2002. China Population Statistics Yearbook 2002. China Statistics Press, Beijing 550pp.

Cifuentes, L., Borja-Aburto, V.H., Gouveia, N., Thurston, G.D., Davis, D.L., 2001. Assessing the health benefits of urban air pollution reductions associated with climate change mitigation (2000–2020): Santiago, Sao Paulo, Mexico City, and New York City. Environmental Health Perspectives 109 (Supplement 3), 419–425.

Daniels, M.J., Dominici, F., Samet, J.M., Zeger, S.L., 2000. Estimating particulate matter-mortality dose-response curves and threshold levels: an analysis of daily time series for the 20 largest US cities. American Journal of Epidemiology 152 (5), 397–406.

Delucchi, M.A., 2000. Environmental Externalities of motor vehicle use in the US. Journal of Transportation Economics and Policy 34 (2), 135–168.

Dockery, D.W., Pope III, C.A., Xu, X., Spengler, J.D., Ware, J.H., Fay, M.E., Ferris, B.G., Speizer, F.E., 1993. An association between air pollution and mortality in six US cities. The New England Journal of Medicine 329 (24), 1753–1759.

Dominici, F., Shepard, L., Clyde, M., 2003. Health effects of air pollution: a statistical review. International Statistical Review 71 (2), 243–276.

EIA, 2004. Annual energy outlook 2004 with projections to 2025. DOE/EIA-0484(2001), Energy Information Agency. http://www.eia.doe.gov/oiaf/aeo/.

EPA, 1997. The benefits and costs of the Clean Air Act, 1970-1990. EPA-410-R-97-002, United States Environmental

Protection Agency Office of Air and Radiation Policy. http://www.epa.gov/airprogm/oar/sect812/.

EPA, 1999. The benefits and costs of the Clean Air Act, 1990-2010. EPA-410-R-99-001, United States Environmental Protection Agency Office of Air and Radiation Policy.

Feng, T., 1999. Controlling Air Pollution in China: Risk Valuation and the Definition of Environmental Policy. New Horizon in Environmental Economics. Edward Elgar, Cheltenham, UK and Northampton, MA, USA 266pp.

Fridley, D.G., 2001. China Energy Databook v. 5.0. Lawrence Berkeley national Laboratory.

HEI, April 2004. Health Effects of Outdoor Air Pollution in Developing Countries of Asia. Special Report 15, Health Effects Institute, Boston, MA.

Hoek, G., Brunekreef, B., Goldbohm, S., Fischer, P., van der Brandt, P.A., 2002. Association between mortality and indicators of traffic-related air pollution in the Netherlands: a cohort study. The Lancet 360, 1203–1209.

Hurley, F., Hunt, A., Cowie, H., Holland, M., Miller, B., Pye, S., Watkiss, P., 2005. Methodology for the Cost-benefit Analysis for CAFE Volume 2: Health Impact Assessment. European Commission, UK http://europa.eu.int/comm/environment/air/cafe/pdf/cba_methodology_vol2.pdf.

Jin, L.B., Qing, Y., Xu, Z., 2000. Relationship between air pollution and acute and chronic respiratory disease in Benxi (in Chinese). Journal of Environment and Health 17 (5), 268–270.

Kan, H., Chen, B., 2004. Particulate air pollution in urban areas of Shanghai, China: health-based economic assessment. Atmospheric Environment 322, 71–79.

Krewski, D.T.B.R., Goldberg, M.S., Hoover, K., Siemiatycki, J., Jerrett, M., Abrahamowics, M., White, W.H., 2000. Reanalysis of the Harvard Six Cities Study and the American Cancer Society Study of Particulate Air Pollution and Mortality: A Special Report of the Institute's Particle Epidemiology Reanalysis Project, Health Effects Institute, Cambridge, MA. http://www.healtheffects.org/Pubs/Rean-ExecSumm.pdf.

Kunzli, N., 2002. The public health relevance of air pollution abatement. European Respiratory Journal 20, 198–209.

Kunzli, N., et al., 2000. Public-health impact of outdoor and traffic-related air pollution: a European assessment. The Lancet 356, 795–801.

Kunzli, N., Medina, S., Kaiser, R., Quenel, P., Horak, F., Studnicka, M., 2001. Assessment of deaths attributable to air pollution: should we use risk estimates based on time series or on cohort studies? American Journal of Epidemiology 153 (11), 1050–1055.

Larson, E.D., Ren, T., 2003. Synthetic fuel production by indirect coal liquefaction. Energy for Sustainable Development 7 (4), 21–44.

Leksell, I., Rabl, A., 2001. Air pollution and mortality: quantification and valuation of years of life lost. Risk Analysis 21 (5), 843–857.

Levy, J.I., Hammitt, J.K., Yanagisawa, Y., Spengler, J.D., 1999. Development of a new damage function model for power plants: methodology and applications. Environmental Science and Technology 33 (24), 4364–4372.

Li, J., Guttikunda, S.K., Carmichael, G.R., Streets, D.G., Chang, Y.-S., Fung, V., 2004. Quantifying the human health benefits of curbing air pollution in Shanghai. Journal of Environmental Management 70 (1), 49–62.

Lvovsky, K., Hughes, G., Maddison, D., Ostro, B.D., Pearce, D., 2000. Environmental costs of fossil fuels: a rapid assessment method with application to six cities. The World Bank.

Ma, H.B., Hong, C.J., 1992. Effects of particulate air pollution on respiratory disease. Chinese Journal of Public Health 11 (4), 229–232.

Ogden, J.M., Williams, R.H., Larson, E.D., 2004. Societal lifecycle costs of cars with alternative fuels/engines. Energy Policy 32 (1), 7–27.

Ostro, B.D., 1987. Air pollution and morbidity revisited: a specification test. Journal of Environmental Economics and Management 14, 87–98.

Ostro, B.D., 1990. Associations between morbidity and alternative measures of particulate matter. Risk Analysis 10 (3), 421–427.

Pope III, C.A., 2000a. Epidemiology of fine particulate air pollution and human health: biologic mechanisms and who's at risk? Environmental Health Perspectives 108 (Supplement 4), 713–723.

Pope III, C.A., 2000b. Invited commentary: particulate matter-mortality exposure-response relations and threshold. American Journal of Epidemiology 152 (5), 407–412.

Pope III, C.A., Burnett, R.T., Thun, M.J., Calle, E.E., Krewski, D., Ito, K., Thurston, G.D., 2002. Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. Journal of American Medical Association 287 (9), 1132–1141.

Rabl, A., 2003. Interpretation of air pollution mortality: number of deaths or years of life lost? Journal of Air and Waste Management Association 53, 41–50.

Rabl, A., Spadaro, J.V., 2000. Public health impact of air pollution and implications for the energy system. Annual Review of Energy and the Environment 25, 601–627.

Rowe, R.D., Lang, C.M., Chestnut, L.G., Latimer, D.A., Rae, D.S., Bernow, S.M., White, D.E., 1995a. The New York Electricity Externality Study Volume I: Introduction and Methods, I. Oceania Publications Inc. 839pp.

Rowe, R.D., Lang, C.M., Chestnut, L.G., Latimer, D.A., Rae, D.S., Bernow, S.M., White, D.E., 1995b. The New York Electricity Externality Study Volume II: Appendices, II. Oceania Publications Inc. 720pp.

Samet, J.M., Zeger, S.L., Dominici, F., Curriero, F., Coursac, I., Dockery, D.W., Schwartz, J., Zanobetti, A., 2000. The National Morbidity, Mortality and Air Pollution Study Part II: Morbidity and Mortality From Air Pollution in the United States. Health Effects Institute, Cambridge, MA.

Samoli, E., et al., 2003. Investigating the dose-response relation between air pollution and total mortality in the APHEA-2 multicity project. Occupational and Environmental Medicine 60 (12), 977–982.

Spix, C., et al., 1998. Short-term effects of air pollution on hospital admissions of respiratory diseases in Europe: a quantitative summary of APHEA study results. Archives of Environmental Health 53 (1), 54–64.

The University of Michigan China Data Center, 2003. The Complete Collection of China 2000 Provincial Population Census Data Assembly http://chinadatacenter.org/newcdc/census2000.htm.

Wang, X., 1997. Comparison of constraints on coal and biomass fuels development in China's energy future. Ph.D. Thesis, University of California, Berkeley, Berkeley, 533pp.

Wang, X., 2004. Evaluating Impacts of Air Pollution in China on Agriculture and Public Health: Implications for Air Pollution and Energy Policies. Ph.D. Thesis, Princeton University, Princeton, NJ.

Wang, H., Mullahy, J., Chen, D., Wang, L., Peng, R., 2001. Willingness to Pay for Reducing the Risk of Death by Improving Air Quality: A Contingent Valuation Study in Chongqing. China. Third International Health Economic Association Conference, York, UK.

Wang, X., Mauzerall, D.L., Hu, Y.T., Russell, A., Larson, E., Woo, J.-H., Streets, D.G., Guenther, A.B., 2005. A high-resolution emission inventory for eastern China in 2000 and three scenarios for 2020. Atmospheric Environment 39, 5917–5933.

Williams, R.H., 2001. Toward zero emissions from coal in China. Energy for Sustainable Development 5 (4), 39–65.

Williams, R.H., Larson, E.D., 2003. A comparison of direct and indirect liquefaction technologies for making fluid fuels from coal. Energy for Sustainable Development 7 (4), 103–129.

Woodruff, T.J., Grillo, J., Schoendorf, K.C., 1997. The relationship between selected causes of postneonatal infant mortality an particulate air pollution in the United States. Environment and Health Perspectives 105, 608–612.

World Bank, 2001. GDP per capita, http://www.worldbank.org/data/databytopic/GDP_PPP.pdf.

Zheng, H., 2003. Case study of coal gasification-based energy supply system in Zaozhuang City. Ph.D. Thesis, Tsinghua University, China, Beijing.

Zheng, H., Li, Z., Ni, W., Ren, T., Larson, E., 2003. Case study of coal gasification-based energy supply system for China. Energy for Sustainable Development 7 (4), 30–38.