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Quantifying the human health benefits of curbing air pollution in Shanghai

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Abstract

Urban development in the mega-cities of Asia has caused detrimental effects on the human health of its inhabitants through air pollution. However, averting these health damages by investing in clean energy and industrial technologies and measures can be expensive. Many cities do not have the capital to make such investments or may prefer to invest that capital elsewhere. In this article, we examine the city of Shanghai, China, and perform an illustrative cost/benefit analysis of air pollution control. Between 1995 and 2020 we expect that Shanghai will continue to grow rapidly. Increased demands for energy will cause increased use of fossil fuels and increased emissions of air pollutants. In this work, we examine emissions of particles smaller than 10 μm in diameter (PM_{10}), which have been associated with inhalation health effects. We hypothesize the establishment of a new technology strategy for coal-fired power generation after 2010 and a new industrial coal-use policy. The health benefits of pollution reduction are compared with the investment costs for the new strategies. The study shows that the benefit-to-cost ratio is in the range of 1–5 for the power-sector initiative and 2–15 for the industrial-sector initiative. Thus, there appear to be considerable net benefits for these strategies, which could be very large depending on the valuation of health effects in China today and in the future. This study therefore provides economic grounds for supporting investments in air pollution control in developing cities like Shanghai.

Keywords: Particulate matter; PM_{10}; Air pollution control; Human health benefits; Shanghai; China

1. Introduction

Air pollution is taking a toll on the health of people living in large cities in the developing world. Economic development and urban growth are coupled with surges in energy consumption to fuel industrial production, the growing transportation demand, and elevated living standards. Because the energy choice is mainly fossil fuels, the air pollutant emissions that are generated have inflicted significant health damages on the population, despite the unarguable benefits brought about by the economic boom. In many cities, the pollution levels are often several times the limits recommended by the World Health Organization (World Health Organization, 2000) to protect human health. Air pollutant species of the greatest concern are particulate matter (PM), sulfur dioxide (SO_2), nitrogen oxides (NO_x), carbon monoxide (CO), ozone (O_3) through secondary formation, and—not to a lesser extent—toxic substances.

The challenge to manage limited economic resources and maintain a sustainable environment is immense. Investments in cleaner technologies, pollution prevention, and emission controls are often outweighed by the short-term needs to boost domestic production and provide job opportunities. Motivated to investigate whether the costs of clean technology investments and pollution controls are too expensive to afford—as developing countries sometimes assert—we have conducted this study in Shanghai, one of the most rapidly growing and prosperous metropolitan areas in China. Our aim is to cast light on the potential economic benefits from improved population health status as air pollution levels decline and to compare these benefits with the expenditures on emission control measures needed to achieve them.

An earlier study conducted by Streets et al. (1999) examined a suite of clean technology scenarios designed to

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reduce emissions from the industrial and power-generation sectors in Shanghai, including the application of emission control technologies, fuel switching, cogeneration, and energy efficiency improvements. That study evaluated the costs and effectiveness of alternative policy options in reducing population exposures to SO_2 and sulfate (SO_4) pollution. It was shown that emission controls on different sources would yield varying levels of effectiveness. Controlling smaller industrial combustion sources was found to yield the greatest exposure reduction, because industrial facilities are located close to residential areas and emissions disperse in the local vicinity. The power sector, on the other hand, showed smaller potential to reduce human exposures, because emissions from power plants tend to disperse in the long range from taller smokestacks.

However, the previous study did not quantify the health improvement potential of the various emission control scenarios; it used as a metric of effectiveness simply the exposure of populations to SO_2 and SO_4 levels in excess of the WHO guidelines. Nor did it examine fine particulate matter—a key pollutant linked with premature mortality, chronic pulmonary disease, and acute lung function impairment—of which SO_4 is one component (Pope et al., 1995; Dockery et al., 1993). For this present study, we select particulate matter of diameter less than 10 μm, or PM_{10}, for an in-depth analysis. Two illustrative air pollution control scenarios are evaluated using dose-response relationships established in epidemiological studies in China and elsewhere in the world. Not much research has been done to understand the human health impacts of PM_{10} and fine particles (such as $PM_{2.5}$) at urban scale in China. One of the reasons is that the monitoring metric of particulate matter has traditionally been total suspended particulates (TSP); measurement of the small PM fractions is lacking. The absence of monitoring data makes it impossible to determine the exposure levels.

It should be noted that at the time this study was being performed, a similar effort was being carried out by local researchers to evaluate energy options and health effects in Shanghai (Chen et al., 2002). That study, which used a different modeling approach to the energy and air quality scenarios and a similar approach to the health benefits analysis, also found significant health benefits—between \$540 and \$887 million per year—from energy efficiency improvements and the expanded use of natural gas to replace coal consumption in the city. Their conclusion is consistent with our analysis.

1.1. Shanghai—pollution and population

Shanghai is one of China's most populous and prosperous metropolitan areas. Situated on the estuary of the Yangtze River facing the East China Sea, the city is home to more than 13 million residents (see Fig. 1). The economic

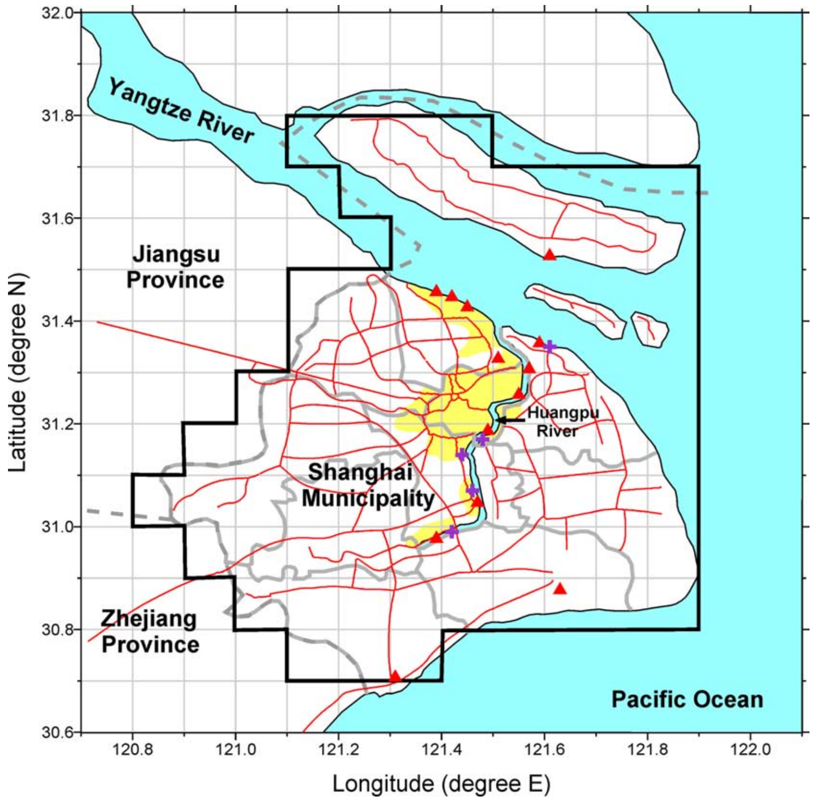

Fig. 1. Map of Shanghai Municipality and modeling domain (solid line indicates 0.1° × 0.1° modeling domain; shaded area is Shanghai metropolitan area; triangles are power plants; and crosses are large industrial facilities).

boom in Shanghai came with greatly increased demand for energy to fuel its growing economy. Shanghai's energy use has traditionally been dominated by coal, which accounted for about 70% of total energy consumption in the early 1990s. Air and water pollution has historically been severe in Shanghai and associated with adverse health effects (Fang, 1989; Chen, 1994). Recently, advancement in economic conditions has nourished the development of environmental regulation and investments in environmental management and pollution controls. In the last 10 years, environmental policies that include relocating heavy industries from the urban centers to less-populated remote areas, replacing residential coal use with natural gas or coal gas, converting coal-fired industrial boilers to cleaner fuels, and installing end-of-pipe emission controls at large power plants have steadily reduced emissions of air pollutants. As shown in Fig. 2, which depicts the recent trends (indexed) in ambient NO_x, SO_2 and TSP concentrations, average SO_2 and TSP concentrations have declined significantly since 1990 in both urban and suburban locations. In contrast, emissions of NO_x have increased dramatically during the same period, as a result of the rapid growth in the number of vehicles. The air quality monitoring data show that by 1999 annual average TSP concentrations in the urban area of Shanghai had dropped to 168 $\mu g/m^3$, a reduction of over 50% from the 1990 level (Shanghai Environmental Protection Bureau, 2000). The mean SO_2 concentration was 44 $\mu g/m^3$ in 1999, below the WHO annual threshold of 50 $\mu g/m^3$ (World Health Organization, 2000).

Nevertheless, current particulate levels in Shanghai are high. (The WHO does not present a threshold for PM levels, suggesting instead that there is no lower limit for health damage from PM.) Moreover, monitoring data in the late 1990s show a geographic redistribution of particulate pollution in Shanghai. The annual average TSP level in Shanghai's downtown area in 1999 was reduced by 8% from 182 to 168 $\mu g/m^3$ from the 1998 level, but the suburban TSP level of 152 $\mu g/m^3$ increased slightly from the 1998 level of 147 $\mu g/m^3$ (Shanghai EPB, 2000). This trend can also be seen in Fig. 2. This reflects a transition in economic activities, as more manufacturing businesses move out of the heart of the city. The demography of Shanghai shows a similar pattern—that more residents are migrating from the city to the suburban areas. Focusing only on reductions in urban pollution may therefore mask the risks of increased population exposures to air pollution in the city's outskirts. For this reason, our study encompasses the entire area of the Shanghai Municipality, not just the downtown area or the county area.

In addition to particulate pollution from stationary sources, the burgeoning vehicle fleet in Shanghai has significantly driven up emissions of NO_x and volatile organic compounds (VOC) in the city. The health hazards associated with NO_x itself include complications in pulmonary diseases and increased susceptibility to respiratory infection. NO_x and VOC are also known to be precursors of ground-level ozone (O_3), a secondary pollutant that is linked with a number of adverse health effects. Exposures to ozone, even at very low levels, can trigger impairment of lung function and development of chronic pulmonary disease. Children and asthma patients are particularly susceptible to ozone pollution. Xu et al.

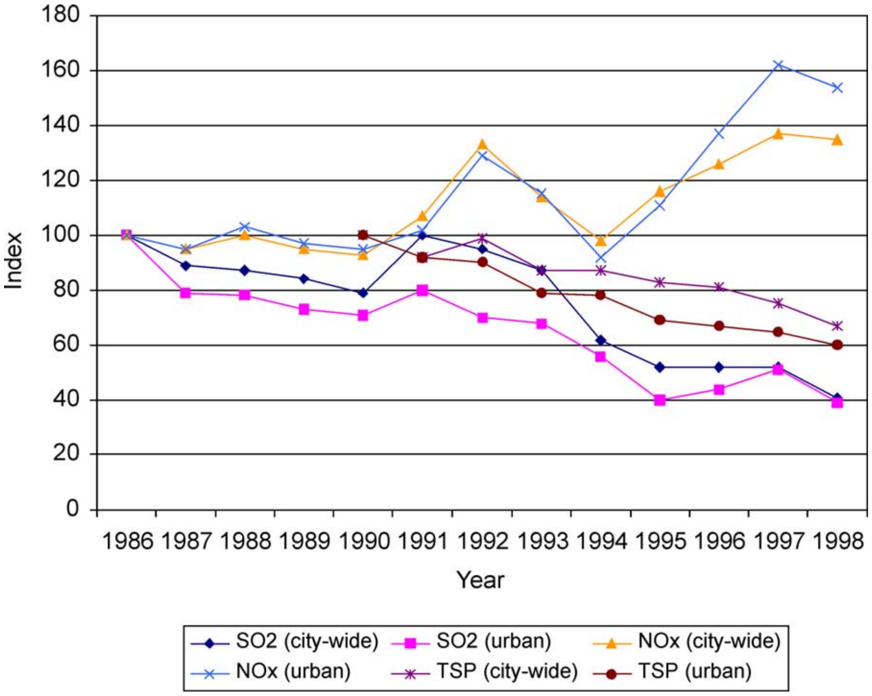

Fig. 2. Indexed trends for annual average SO_2, NO_x and TSP concentrations in Shanghai between 1986 and 1998 (city-wide refers to the city averages and urban refers to the downtown area).

(1999) pointed out that the prevailing climatological conditions in Shanghai are conducive to ozone production.

We have quantified the NO_x emission changes associated with transportation and stationary fuel combustion sources in Shanghai, but we do not at present have a satisfactory NO_x or O_3 atmospheric chemistry model that is parameterized for Shanghai; and so quantification of the benefits of controlling vehicular emissions must await future model development. Nevertheless, the increased pollution from mobile sources is a great concern for Shanghai. Technical and policy measures should be adopted to address fuel quality, vehicle efficiency, tailpipe controls, urban planning, and land-use policy. Concern has also been expressed about the associated emissions of greenhouse gases and the difficulty of holding down CO_2 emissions in Shanghai in the future (Wu et al., 1997; Gielen and Chen, 2001).

2. Methodology

This study investigates the implications of reduced particulate matter (PM_{10}) pollution for the health of Shanghai inhabitants. We examine the situation in 1995 and select the year 2020 to compare the policy options associated with a business-as-usual (BAU) pathway and two alternative control scenarios. Particulate matter is a mixture of solid particles and liquid droplets in the air, which originate from fossil-fuel combustion, other anthropogenic activities, and natural sources. The particles may be primarily carbonaceous in nature, if they originate from combustion activities, or mineral. Fine particles ($PM_{2.5}$) are characterized as particles with a diameter of 2.5 μm or less, and coarser particles (PM_{10}) are those that are 10 μm in diameter or less. $PM_{2.5}$ primarily comes from fuel combustion sources (i.e. power plants, industrial facilities, mobile sources, and residential heating and cooking stoves); it includes aerosol conversion products such as sulfate (SO_4) and nitrate (NO_3). PM_{10} can come from fuel combustion, other anthropogenic sources like cement manufacture, and natural sources such as road dust, construction dust, and windblown soil.

In general, fine particles ($PM_{2.5}$) are a better surrogate than PM_{10} for the mixture of particulate matter from fuel combustion sources, and they are of greatest health concern because they penetrate deeper into the lung and tend to be more reactive. The health literature has used both $PM_{2.5}$ and PM_{10} measures to investigate the relationship between particulate pollution and health impacts; however, PM_{10} studies are more prevalent because PM_{10} concentrations are much easier to measure in the ambient air. The health effects of particulate matter pollution are discussed in more detail later in this paper. Considering the wealth of epidemiological studies linking PM_{10} with various health endpoints (i.e. mortality, hospital visits, and emergency room visits), PM_{10} is selected as the parameter for evaluating the health benefits in this study. Though existing studies suggest that the severity of the health effects of $PM_{2.5}$ pollution is much higher than that of the PM_{10} pollution (Schwartz et al., 1996), this study employs the PM_{10} pollution indicator because of the nature of the available data and the measurements that have been made by the Shanghai Environmental Protection Bureau (EPB).

This study takes an integrated approach that links the emission sources, the ambient pollution distributions, and the human exposures to particulate matter in Shanghai. Two scenarios are developed to illustrate the control options for emissions from two of the key energy sectors in Shanghai: power generation and industry. The emission inventories, air dispersion modeling, and health impact analysis are developed within a common spatial framework at $0.1° \times 0.1°$ grid resolution of the Shanghai city area. The emissions, atmospheric dispersion, and transformation of the PM_{10} species components are estimated separately and aggregated to derive total PM_{10} emissions and concentrations. The species include primary carbonaceous PM, primary mineral PM, as well as the secondary sulfate formed in the atmosphere from SO_2 emissions. Secondary nitrate is not included in this analysis because there is no appropriate modeling tool available for Shanghai with which to estimate it. However, composition analysis for $PM_{2.5}$ in Shanghai (Yao et al., 2002) shows that the contribution of nitrate in the $PM_{2.5}$ mixture is still relatively small in Shanghai. According to chemical analysis, the fractions of water-soluble ions (excluding dust) in $PM_{2.5}$ are 46% sulfate, 18% nitrate, and 17% ammonium. Nevertheless, it should be noted the role of NO_x emissions in the formation of nitrate and ozone is becoming increasingly important in Shanghai and should be a priority for future study.

2.1. Energy and emissions profiles

Energy use in Shanghai is based on data in the latest version of the RAINS-Asia model (Version 8). RAINS-Asia is an analytical tool designed to investigate acid deposition and ambient air quality in Asia (Foell et al., 1995; Shah et al., 2000; Guttikunda et al., 2003). Detailed socio-economic and energy data have been gathered for 23 Asian countries and disaggregated into 94 sub-regions. Twenty-four of the sub-regions are large metropolitan areas in Asia, and Shanghai is one of them. The dataset provides parameters needed to compute energy use by fuel type and by energy-use sector. Emissions are calculated based on the actual or standard emission rates by fuel and combustion technology. The year 1995 is selected as the base year to establish energy and emission baselines, which are then projected out to a 25-year time horizon to present a long-term view of the development of Shanghai, using the RAINS-Asia model. Emissions from non-energy activities are estimated from activity data contained in local and national statistical yearbooks. Emissions of particles from road dust were estimated from work by Ji et al. (1993). The distribution of small industrial facilities was taken from Zhao and Zhao

(1985), and we implicitly assume that this distribution did not change much prior to 1995. Emissions were allocated to grid cells using appropriate distributions of total and rural population, small industrial facilities, road traffic, and rivers and ocean lanes. The 18 largest point sources (13 power plants and five industrial complexes) were placed at their latitude/longitude coordinates (see Fig. 1); their emissions were calculated individually and subtracted from the total.

2.2. Air pollution dispersion modeling

This study uses the Urban Branching Atmospheric Trajectory Model (UrBAT) to calculate the ground-level concentrations of SO_2, PM_{10}, and $PM_{2.5}$. UrBAT is a three-layer Lagrangian puff-transport forward trajectory model (Calori and Carmichael, 1999). The model has detailed meteorological input data, including 6-h precipitation and wind vector data for the year 2000 (NCEP, 2000). The study domain extends from 120.7°E to 122.0°E in longitude and 30.7°N to 32.0°N in latitude covering an area of about 13,000 km^2 (see Fig. 1). The area of the Shanghai urban center is about 3200 km^2 and of the entire Shanghai Municipality about 6300 km^2. Pollution generated outside the study domain and imported into the domain is calculated using the full ATMOS modeling component for the whole of Asia, as documented in Foell et al. (1995) and Shah et al. (2000).

2.3. Health benefit analysis

A risk assessment method is used to evaluate the human exposure hazards to particulate matter and the health benefits from reduced exposure levels. The health status is measured at various endpoints (e.g. acute mortality, chronic pulmonary disease, hospital visits, and emergency room visits). Established dose-response functions from epidemiological studies conducted in China and other countries are used to estimate the changes in health status from reduced particulate matter pollution levels. The benefits of reduced health risks are then translated into economic terms by applying economic valuation methods (e.g. willingness-to-pay, cost-of-illness) and compared with the costs of the alternative emission reduction measures.

3. Results

3.1. Energy use

In 1995, according to the RAINS-Asia model, energy consumption in Shanghai totaled 1077 PJ, of which 50% was met by coal (Table 1). Derived coal, like coke and briquettes, supplied another 15% of total energy use. Petroleum products provided about 25% of the total energy, of which heavy fuel oil accounted for 47%, light fraction fuels (e.g. gasoline, kerosene, and liquefied petroleum gas) accounted for 31%, and medium distillates (e.g. diesel) accounted for 23%. Natural gas accounted for only 9% in the energy supply picture in 1995. The industrial sector was the single largest energy-consuming sector in 1995, using about 50% of the total energy (Table 2). Power generation used about one-third of the energy; the transport and domestic energy use was relatively small, accounting for 10 and 8% of total energy consumption, respectively. Future energy consumption in Shanghai is determined by several factors, including forecasted economic growth, sectoral shifts, and population growth, as embedded in the RAINS-Asia model.

The population in Shanghai Municipality was 14.2 million in 1995 (China Statistical Yearbook, 1996), of which 13 million lived in the downtown metropolitan area. To estimate the population of the Municipality in 2020, we used growth rates from the United Nations (United Nations, 1998) out to 2015 and an assumed annual average growth rate of 1% between 2015 and 2020. Thus, the total population of the Shanghai Municipality in 2020 is forecast to be 19.7 million in this study. This is somewhat higher than the 2020 population assumed in the RAINS-Asia model (16.9 million), so the energy generation and emissions may be relatively lower, which would understate the benefits we find. Fig. 3 shows the assumed population distribution in the greater Shanghai Municipality in 1995 and 2020. The spatial allocation scheme for population is derived from Tobler et al. (1995).

Based on the above assumptions, it is projected that total energy use in Shanghai will rise to 1758 PJ in 2020, an increase of 63% from the 1995 level. The increase is largely met by coal. Table 1 shows the projected energy

Table 1
Energy consumption in Shanghai Municipality in 1995 and 2020

Fuel type	1995 (PJ)	2020 (PJ)
Hard coal	532.2	932.0
Derived coal	163.9	215.9
Heavy fuel oil	124.7	194.3
Medium distillates	60.5	139.4
Light fractions	81.5	100.8
Natural gas	100.4	158.5
Solid waste (inc. biomass)	13.8	16.7
Renewables	0.0	0.0
Total	1076.9	1757.5

Table 2
Energy consumption by economic sector in Shanghai in 1995 and 2020

Sector	1995 (PJ)	2020 (PJ)
Power	327.7	781.7
Industry	554.3	573.2
Domestic	83.8	94.2
Transport	111.1	308.4
Total	1076.9	1757.5

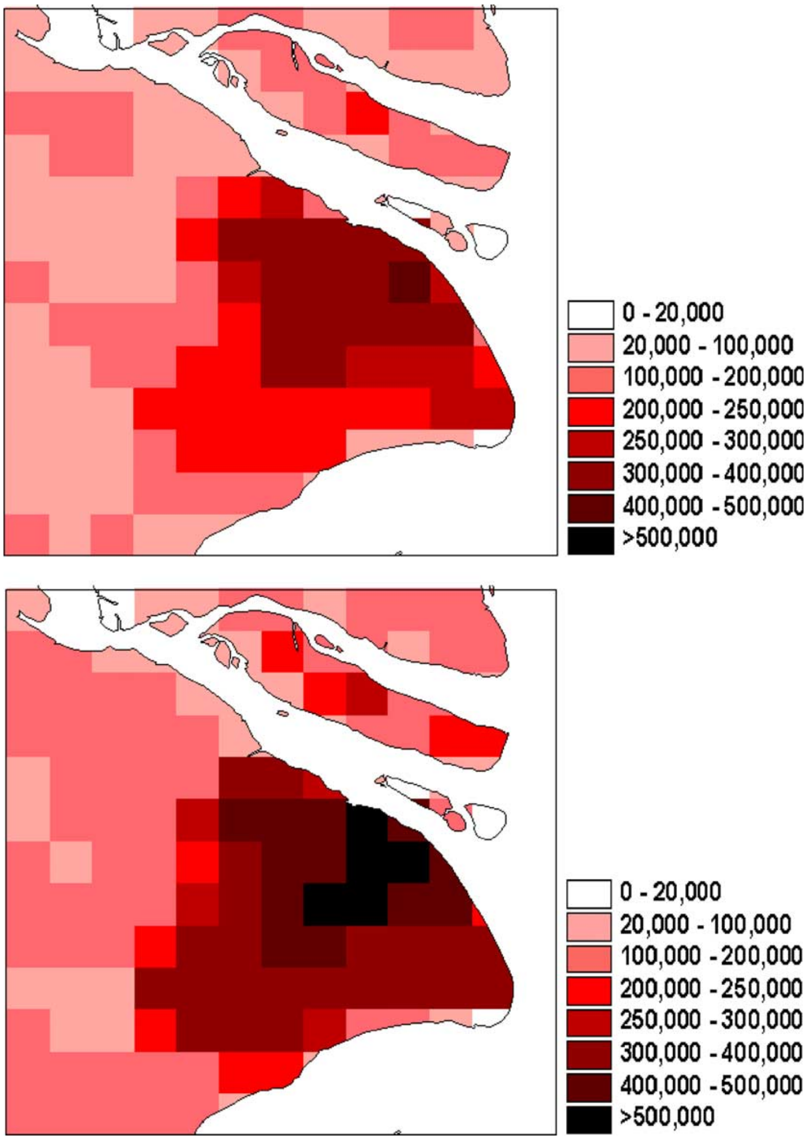

Fig. 3. Population distribution in Shanghai in 1995 (upper) and 2020 (lower).

use by fuel type for 2020. The forecasts from the RAINS-Asia model were made before the contribution from the Three Gorges hydroelectric project was known; the Three Gorges project clearly has the ability to contribute to alleviating air pollution in Shanghai if indeed electricity from that project is ultimately delivered to Shanghai. No contribution from nuclear power is expected in this time frame. The energy-use pattern reflects a shift among the energy-consuming sectors over the next 25 years, as shown in Table 2. Most noticeably, power generation becomes the largest energy-use sector in 2020, accounting for 44% of total energy use. Industrial fuel use is projected to remain at roughly the 1995 level, reflecting the trends of closing down heavy industry in the city and continuing to improve the efficiency of energy use in industrial facilities. Energy use in the transportation sector grows almost threefold between 1995 and 2020.

3.2. Pollution control scenarios and costs

The study formulates two illustrative scenarios to control emissions from two key energy-use sectors in Shanghai: power generation and industrial manufacturing. The control measures adopted under each of the scenarios are described below.

Control scenario for power generation (C1). This scenario assumes that all new power generation capacity coming on line in the period 2010–2020 will adopt state-of-the-art integrated gasification combined cycle (IGCC) technology for coal combustion. This amounts to a total generating capacity equivalent to 366 PJ, in addition to the planned new capacity of natural gas and oil. The performance characteristics for IGCC in this analysis are based on the intermediate oxygen-blown DESTEC ‘G’ process in a 400 MW combustion turbine unit manufactured by Westinghouse (USDOE, 1998). It is assumed that the new IGCC plant has a net heat rate of 7513 Btu/kWh with 45.5% efficiency. The levelized cost of power generation of the IGCC system is 3.76 cents/kWh and the total annual costs of the generating plants are estimated to be $1.74 billion in 2020.

In the business-as-usual case, it is assumed that conventional pulverized-coal plants would fuel the electricity demand. We assume that the largest of these plants would install flue-gas desulfurization (FGD) units to control SO_2 emissions and the rest would use low-sulfur coal, as forecast by the RAINS-Asia model. The incremental costs of the C1 Scenario are thus the difference between the IGCC investments and the avoided investments in these conventional coal plants. The levelized costs for the conventional plants without any control measures are estimated to be 2.80 cents/kWh. At the lower heat rate of 9077 Btu/kWh, the costs for operating these conventional coal plants without any control measures would amount to $1.29 billion for Shanghai in 2020. From the RAINS-Asia model, the control measures amount to $48 million. Thus, the annual costs of Scenario C1 are $395 million.

Control scenario for industrial coal use (C2). This scenario assumes that all industrial coal use is banned in the Shanghai urban area by 2020. It is further assumed that 75% of the coal-fired production activities will simply close down and not be replaced. The remaining 25% will be relocated to four neighboring counties: Baoshan, Jiading, Shanghai, and Chuansha. This control scenario is consistent with the urban development policy Shanghai is currently pursuing—namely that heavy-polluting industries are moved away from the densely populated urban area. Total industrial coal use in the metropolitan area of Shanghai in 2020 is estimated to be 172 PJ, and therefore the industrial coal use relocated to suburban counties amounts to 43 PJ.

On the cost side, it is assumed that the 75% of industrial coal use closed down in the metropolitan area will not be replaced and that no net cost will be incurred from decommissioning. Based on current practice in China, it is common that equipment from shut-down facilities in the cities gets exported to neighboring counties and re-used or dismantled and used for parts. The facility buildings will be re-used for other purposes. Therefore, there are limited 'stranded' costs associated with plant shutdowns. We assume that the only costs of the C2 Scenario are associated with the 25% of coal use that is relocated and replaced in the form of new industrial boilers installed in existing facilities in the four neighboring counties. The modern coal-burning boiler configuration used in China is standard 10 tons/h, modular, water-tube boilers using traveling-grate stokers for coal combustion (World Bank, 1996). The average price for such boilers is $54,400 in 1995 value. The operating costs are assumed to be equal to the operating costs of the replaced facilities. Assuming operation of 50 h/week and 50 weeks per year, the total annual cost of replacement is estimated to be $94 million in 2020. It should be noted this industrial cost analysis focuses on the fate of coal-consuming industries and does not attempt to capture the macroeconomic impacts of the measure, such as changes in employment, productivity, and gross domestic products (GDP). However, given the dynamic growth of the region, it is quite likely that lost production and lost jobs will be regained in new, modern plants burning oil or gas. This would not affect the air pollution calculations of this study and would minimize any unaccounted-for dislocation costs.

3.3. Emissions of multiple pollutants

An emission inventory is developed for each of the major pollutants: SO_2, NO_x, PM_{10}, $PM_{2.5}$, and TSP for four scenarios: 1995 base year, 2020 business-as-usual (BAU) forecast, and 2020 C1 and C2 control scenarios. Emissions are estimated for each economic sector based on the use of each fuel type. A set of emission rates is developed for each species based on typical emission rates for Chinese facilities (Streets and Waldhoff, 2000). For the new technology options, for which China-specific emission data are not available, standard emission rates derived from western experience are adopted (USDOE, 1998; USEPA, 2001). Table 3 presents the baseline emissions of multiple pollutants from the different sectors in 1995 and the projected emissions in 2020 under the BAU scenario and the two control scenarios. These emission estimates are consistent with the work of Gielen and Chen (2001). Note that we assume continued improvement in particulate emission levels over time, even under a BAU situation. While this trend is developed from the modeling of technology replacement, it is also consistent with local policy to improve the urban environment of Shanghai. Emissions of SO_2 are forecast to rise under the BAU Scenario. Though we recognize that considerable efforts are underway by local authorities to hold SO_2 levels constant through application of emission controls, it is not clear that this can be easily achieved once the cheaper options have been implemented and economic growth picks up again. NO_x emissions are forecast to rise considerably because of growth in the transport sector. Fig. 4 presents the spatial distribution of emissions of PM_{10} under the BAU Scenario in 1995 and 2020.

Compared to the 2020 BAU case, imposing the control scenario C1 on the power sector would reduce 2020 emissions of SO_2, PM_{10}, $PM_{2.5}$, and NO_x by 41, 6, 6, and 13%, respectively. The industrial control scenario C2 results in more significant reductions in particulate emissions: 18% in PM_{10} and 16% in $PM_{2.5}$. However, the reductions in SO_2 and NO_x emissions are less than for C1: 14 and 6%, respectively. Thus, we see the complex interplay between species when different technology options are chosen. The advantage of our method is that we are able to assess the combined effects of these reductions on human health.

Table 3
Summary of emission estimates for 1995 (base year) and 2020 (future year) with and without emission controls

Scenario/source category	Annual emission rates (kt)[a]							
	SO_2	NO_x	TSP/C	PM_{10}/C	$PM_{2.5}$/C	TSP/M	PM_{10}/M	$PM_{2.5}$/M
1995								
Power	214.07	80.42	60.12	40.57	18.07	0	0	0
Industry	199.85	71.12	70.5	49.19	18.3	62.9	31.45	9.02
Residential/commercial	31.89	5.87	20.91	10.41	6.84	0	0	0
Transport	11.58	125.84	10.39	10.12	5.97	0	0	0
Other[b]	1.03	2.51	7.73	6.95	5.88	65.9	18.02	4.63
Total	458.42	285.76	169.65	117.24	55.06	128.8	49.47	13.65
2020B[c]								
Power	394.25	112.74	17.68	11.15	5.05	0	0	0
Industry	214.15	73.23	74.5	52.11	19.58	37.17	18.59	5.33
Residential/commercial	16.82	5.42	10.37	5.23	3.6	0	0	0
Transport	32	276.56	32.01	31.06	16.72	0	0	0
Other	0	0	0	0	0	141.89	36.4	9.25
Total	657.22	467.95	134.56	99.55	44.95	179.06	54.99	14.58
2020P[c,d]								
Power	123.02	50.59	9.36	5.53	2.56	0	0	0
Total	385.99	405.8	126.24	93.93	42.46	179.06	54.99	14.58
Emission reduction (= 2000B-2020P)	271.23	62.15	8.32	5.62	2.49	0	0	0
Emission reduction to 2020B (%)	41	13	6	6	6	0	0	0
2020I[c,d]								
Industry	122.66	46.84	47	33.99	12.17	37.17	18.59	5.33
Total	565.73	441.56	107.06	81.43	37.54	179.06	54.99	14.58
Emission reduction (= 2000B–2020I)	91.49	26.39	27.5	18.12	7.41	0	0	0
Emission reduction to 2020B (%)	14	6	20	18	16	0	0	0

[a] C, carbonaceous, and M, mineral.
[b] Other category includes field combustion, road and construction dust (mineral).
[c] 2020B = Business-As-usual in 2020, 2020P = Power Sector Control in 2020, and 2020I = Industrial Sector Control in 2020.
[d] Only the source category with emission changes is presented.

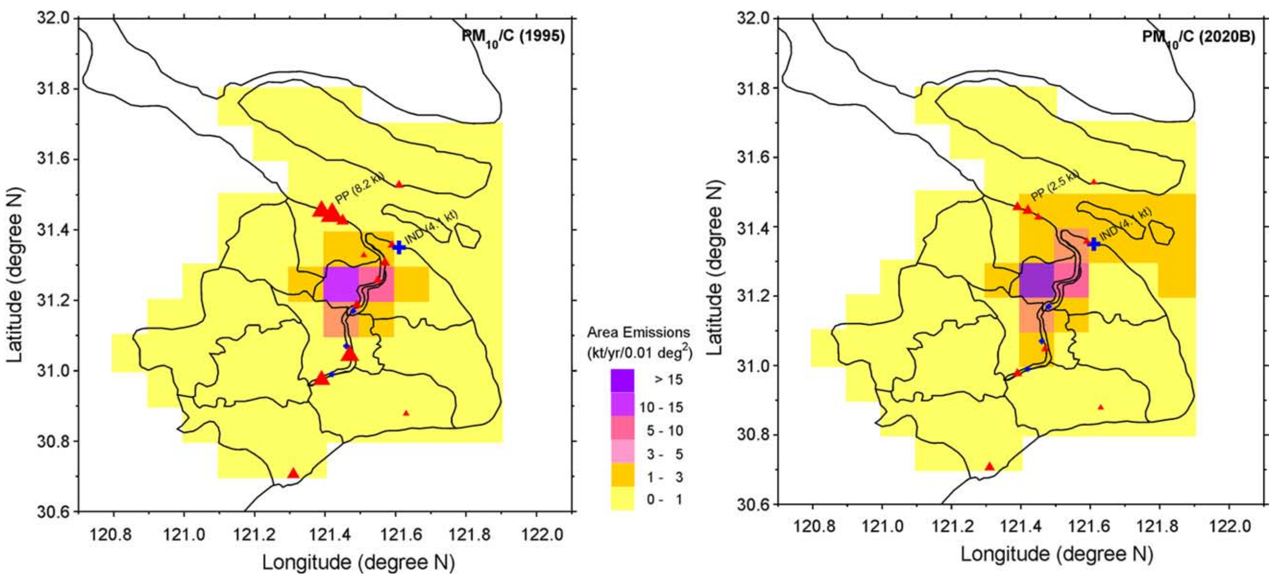

Fig. 4. Annual emission distribution for PM_{10}/C for 1995 and 2020B.

3.4. Atmospheric modeling

The UrBAT model simulations are conducted at a resolution of $0.1° \times 0.1°$ over the greater Shanghai area presented in Fig. 1. The model simulates the atmospheric dispersion patterns and the ambient concentrations of PM_{10} and $PM_{2.5}$ as well as gaseous SO_2. Each component of PM_{10} and $PM_{2.5}$ (primary carbonaceous and mineral PM and secondary sulfate) is simulated separately. The components are then aggregated to yield the total concentrations of PM. Due to differences in the physical and chemical characteristics of PM_{10} and $PM_{2.5}$, particulates were simulated in two separate bins: one containing particles with aerodynamic diameters between 2.5 and 10 μm ($PM_{10\text{-}2.5}$), and the other containing particles smaller than 2.5 μm ($PM_{2.5}$). The $PM_{10-2.5}$, $PM_{2.5}$ and secondary sulfate concentrations, calculated separately, are added together to obtain PM_{10} concentrations for each grid cell. The pollutant concentrations were calculated on a 1-h basis; and the monthly, seasonal, and annual mean concentrations were derived for each grid. The particle size distributions by source type are consistent with the numerical modeling of particulate matter in Shanghai reported by Xu et al. (1992).

The PM modeling results for 1995 and 2020 under the BAU Scenario are presented in Fig. 5. Results of the modeling simulation show that in 1995 the mean annual PM_{10} concentrations ranged from 9 to 121 μg/m^3 with an average of 47 μg/m^3 in the Shanghai Municipality area. The concentrations for SO_2 (not shown) ranged from 4 to 157 μg/m^3, with an average of 53 μg/m^3. In 2020, PM_{10} concentrations increase under the BAU Scenario, ranging from 11 to 133 μg/m^3 with an average of 56 μg/m^3. The eastern area of the Municipality shows an increase in PM_{10} levels of over 30%. SO_2 concentrations range from 5 to 256 μg/m^3, with an average of 72 μg/m^3.

Under the power sector control scenario (C1), pollutant concentrations range from 3 to 142 μg/m^3 for SO_2 and 7–110 μg/m^3 for PM_{10}, with a reduction in concentrations as great as 36 μg/m^3 for PM_{10} relative to the BAU scenario. Similarly, under the industrial control scenario (C2), pollutant concentrations range from 5 to 252 μg/m^3 for SO_2 and 10–104 μg/m^3 for PM_{10} with a reduction in concentrations as great as 35 μg/m^3 for PM_{10}. Fig. 6 presents the average concentrations for PM_{10}, and Fig. 7 presents the predicted reductions in PM_{10} concentrations under the C1 and C2 options in 2020, as compared with the BAU Scenario.

Fig. 7 suggests that the C1 Scenario would have its effects primarily to the north of the city center—on the shoreline of the Yangtze River where several large power plants are located—and to the south of the city, where several smaller power plants are located on the lower reaches of the Huangpu River (see Fig. 1). In contrast, the C2 Scenario mainly affects the central city, where the bulk of industrial coal combustion currently takes place.

3.5. Health implications of the control scenarios

These calculated changes in PM_{10} concentrations are superimposed on the population distribution, and the human health effects calculated. In this study we assume a linear relationship of health response to air pollution at each health endpoint. The formula for estimating changes in health status due to a reduction in an air pollutant is as follows:

$$\Delta E_{i,j} = \Delta P_j \times \beta_{i,j} \times \mathrm{A}_i \times \mathrm{Pop}$$

where

$\Delta E_{i,j}$ = change in incidence cases of health endpoint i during period j

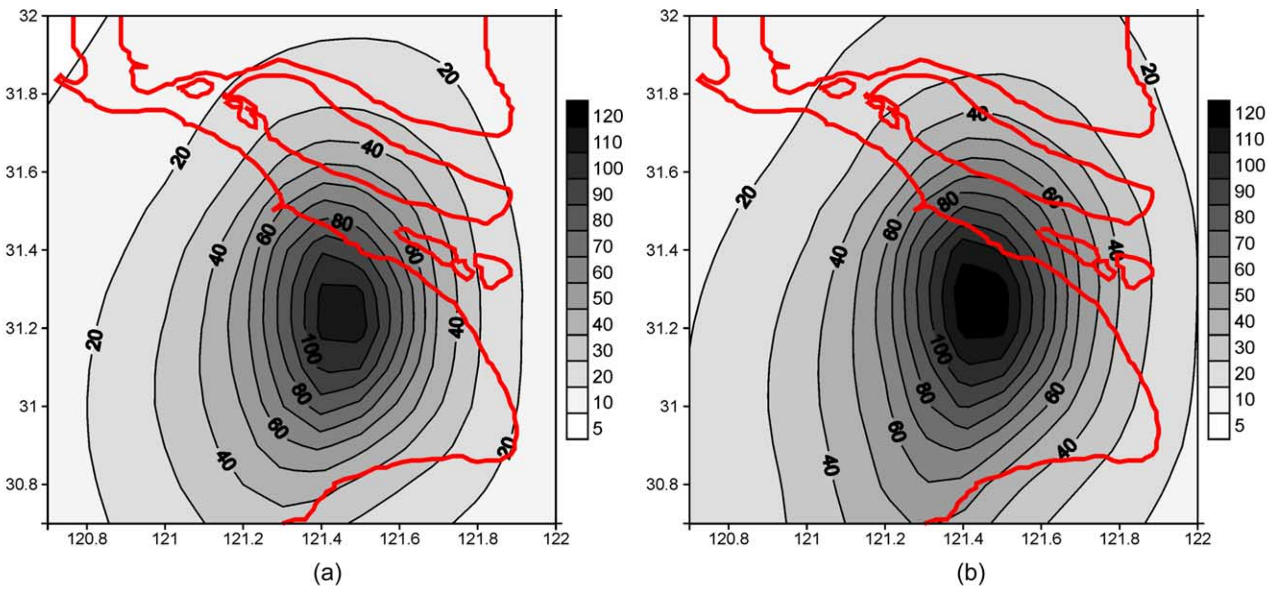

Fig. 5. Annual average PM_{10} concentrations (μg/m^3) in the BAU Scenario for (a) 1995 and (b) 2020.

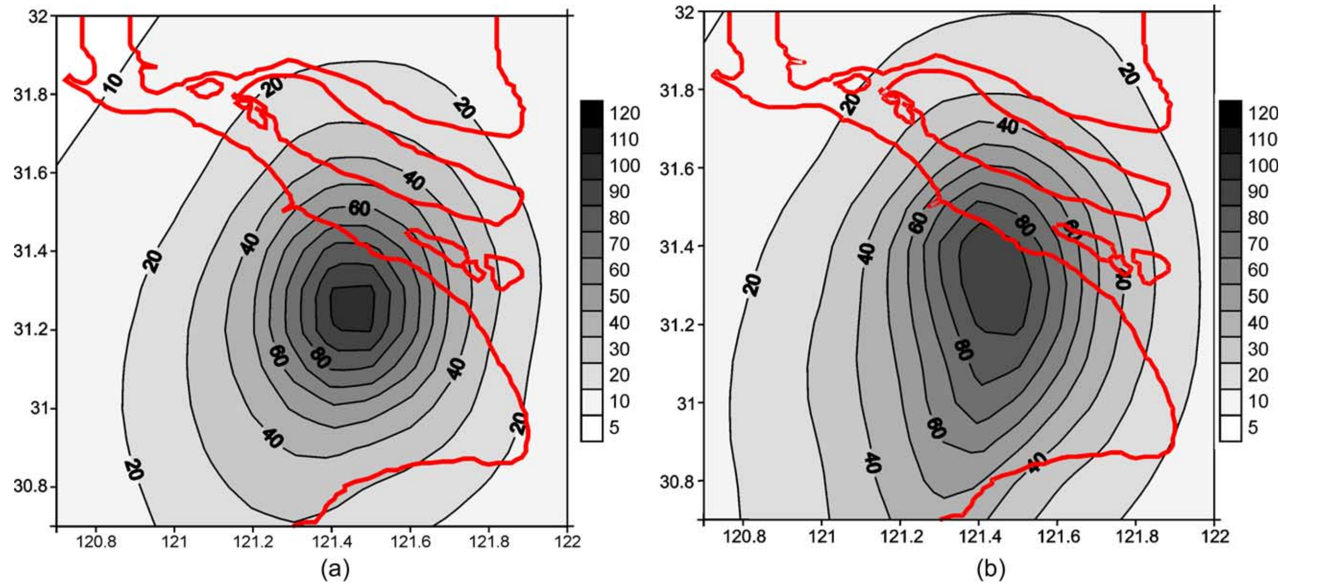

Fig. 6. Annual average PM_{10} concentrations ($\mu g/m^3$) for 2020 in the Control Scenarios (a) Power (C1) and (b) Industry (C2).

ΔP_j = change in pollutant concentration in period j
$\beta_{i,j}$ = dose-response coefficient: the change in incidence rate of health endpoint i in response to unit change in pollutant concentration during period j
A_i = baseline incidence rate of health endpoint i (i.e. mortality, chronic bronchitis) among the population
Pop = population of study region

The following assumptions are applied in this analysis: (1) that the human health responses to air pollution among Shanghai residents are similar to those of residents in other Chinese cities (e.g. Beijing) where the cited epidemiological studies were performed; (2) that the pollution pattern and population baseline health status in Shanghai are similar to those in these other cities; and (3) that the public health status of Shanghai people in 2020 remains the same as in the baseline year for which health data are selected (1998 in this case). A set of dose-response coefficients derived from epidemiological studies is selected. This analysis chooses epidemiological studies conducted in Chinese cities, complemented with studies in the US and other countries where Chinese studies are not available. Table 4 presents the dose-response coefficients used in this analysis.

Epidemiological studies conducted in different countries consistently validate the significant association between PM_{10} pollution and mortality. However, studies that use different methodologies (i.e. different modeling tools or different statistical treatment of data) often do not agree on

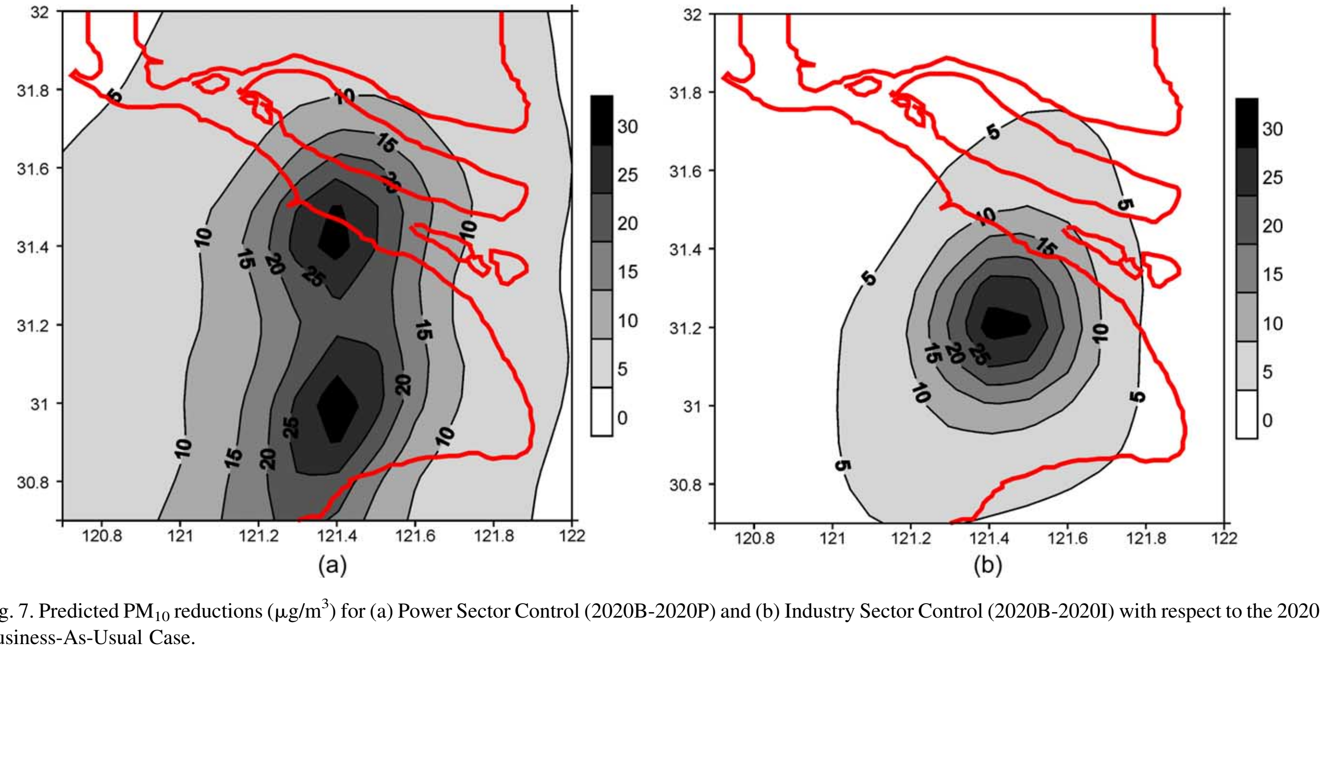

Fig. 7. Predicted PM_{10} reductions ($\mu g/m^3$) for (a) Power Sector Control (2020B-2020P) and (b) Industry Sector Control (2020B-2020I) with respect to the 2020 Business-As-Usual Case.

Table 4
Central estimates for percent change in mortality and morbidity per 10 μg/m^3 increase in annual average PM_{10} concentration

Health endpoint	Change (%)
Mortality	0.84[a]
Hospital visit	0.18[b]
Emergency room visit	0.1[b]
Hospital admission	0.8[c]
Chronic bronchitis	0.1[d]

[a] Lvovsky et al. (2000) (meta-analysis).
[b] Xu et al. (1995) (Beijing, China).
[c] Dockery and Pope (1994) (meta-analysis).
[d] Xu and Wang (1993) (Beijing, China).

the magnitude of the effect. For example, in China most epidemiological studies use TSP as the measure and do not find statistical significance between TSP and mortality (Xu et al., 1994, 2000), whereas the effect of SO_2 on mortality is reported to be more pronounced. Epidemiological studies in industrialized nations often report more significant linkages between particulate matter pollution and mortality and a weaker association for SO_2. Because of the possible variations in dose-response relationships in different geographic regions—due to differences in the concentration levels of pollutants and the public health status of the populations being studied—this work adopts a coefficient based on the results of a meta-analysis (Lvovsky et al., 2000) of nine PM studies, in order to enhance the robustness of the results. Those studies included ones performed in the US; in Santiago, Chile; in Bangkok, Thailand; and in Mexico City, Mexico. The pooled estimate of the weighted-average dose-response coefficient suggests 0.84% change in mortality risk in response to a 10 μg/m^3 change in PM_{10} level.

The morbidity endpoints studied include hospital visits, emergency room visits, and chronic bronchitis. The dose-response coefficients are based on studies conducted in Chinese cities that used TSP as the measure (Xu et al., 1995; Xu and Wang, 1993). Significant associations between daily hospital outpatient and emergency room visits with TSP exposures were repeatedly reported in Beijing, both at high and low air pollution levels. A 100 μg/m^3 increase in TSP was found to be associated with a 1.1% and 0.6% increase in hospital outpatient and emergency room visits, respectively, in Beijing (Xu et al., 1995). We convert the coefficients for PM_{10} based on a chemical analysis of the PM composition of Shanghai's ambient air which shows a PM_{10}/TSP ratio of about 0.6 in the city (Chen et al., 2000).

There is no well-established dose-response relationship linking hospital admissions with air pollution in China. In this analysis, we adopt the results of a meta-analysis of various US studies by Dockery and Pope (1994), which reported a weighted mean of 0.8% increase in hospital admissions with a 10 μg/m^3 increase in daily mean PM_{10}.

Chronic bronchitis is used as an indicator of air-pollution-induced chronic respiratory illness. A cross-sectional study conducted in Beijing to explore the association of indoor and outdoor particulate levels with chronic respiratory illness (Xu and Wang, 1993) reported that a 100 μg/m^3 increase in outdoor TSP pollution levels was associated with an increased risk of bronchitis (odds ratio = 1.9, confidence interval = 1.1–3.2). A study by Schwartz (1993) confirmed the magnitude of chronic bronchitis risk for TSP in the US.

Changes in each of the health endpoints are calculated by applying the dose-response coefficients to the population in Shanghai across the $0.1° \times 0.1°$ grid. Table 5 summarizes the estimates of avoided cases of health impairment at each endpoint for the power and industry control scenarios. As shown in the table, the Power Scenario C1 is estimated to yield 2808 cases of avoided mortality; 96,293 cases of avoided hospital visits; 43,482 fewer cases of hospital admissions; and 1753 fewer cases of chronic bronchitis. In comparison, the Industry Scenario C2 results in 1790 cases of avoided death; 61,379 fewer cases of hospital visits; 27,716 cases of avoided hospital admission; and 1117 fewer cases of chronic bronchitis.

3.6. Economic benefits of improvements in health

As better air quality is proven to reduce the incidence of life-loss, disease, and confinement of activities, the overall well-being of the affected population is improved. We further explore the potential benefits of health improvements among Shanghai residents in economic terms. Both the willingness-to-pay (WTP) estimates and the opportunity-costs-of-illness approach are used to estimate the economic benefits of health improvements. Although much debate surrounds the economic valuation methodology, we believe the exercise can provide useful information on the relative costs and benefits of different mitigation strategies (e.g. controlling industrial vs. power-plant emissions).

Mortality. The willingness-to-pay (WTP) approach is used to estimate the benefits associated with avoided mortality risk from improved air quality. The US Environmental Protection Agency (USEPA, 1997) suggests a median value of statistical life (VOSL) of $4.2 million, based on peer review of the valuation literature of wage-premium trade-offs in the job market. By comparing the purchasing power parity (PPP) of the US and China developed by World Bank (2001), the VOSL is estimated to be $445,000 in China using a direct monetary conversion.

Table 5
Avoided health incidence under the emission control scenarios

Health endpoint	Power scenario C1 (no. of cases)	Industry scenario C2 (no. of cases)
Mortality	2808	1790
Hospital visit	96,293	61,379
Emergency room visit	48,506	30,918
Hospital admission	43,482	27,716
Chronic bronchitis	1753	1117

Though the WTP approach has been widely used in the US for measuring people's attitude toward changes in health risks, transferring the literature from one country to another is not without problems. Individuals' utility functions and the willingness-to-pay for risk avoidance may be significantly different due to social, economic, and cultural factors. In 1998, a contingent valuation survey was conducted in Chongqing to reveal people's willingness-to-pay for reducing the risk of death through improving air quality (Wang et al., 2001). The survey was conducted through a face-to-face interview survey of 550 individuals in the urban residential area in Chongqing, one of the most polluted cities in China. The respondents were given a hypothetical scenario description, followed by open-ended bidding game questions. The results showed that 96% of them answered WTP questions, of whom 62% were able to express their WTP. The study estimated that the range of WTP to save a statistical life was in the range of $16,867 (at 25% probability) to $78,072 (at 75% probability). The median VOSL of the surveyed population was $34,458. Due to the income differential between the surveyed population in Chongqing and the population in Shanghai, we adjust the VOSL value for the Shanghai situation. Wang et al. (2001) concluded that the marginal effect of income on WTP for VOSL is $14,434 for every 100 Yuan of income differential. The average income of the Shanghai population is about four times of that of the surveyed population. Thus, the estimated VOSL of the Shanghai population is approximately $150,000 after adjusting the Chongqing WTP result with income differential. The World Bank (1997) has suggested a VOSL of $60,000 in China. We use this number as the lower end of the range of VOSL estimates. The three values ($60,000, $150,000, and $445,000) used in this analysis establish a range of economic values for reduced mortality risks. The economic benefits of avoided mortality are estimated to range from $139 to $1,030 million in the case of Scenario C1 and from $88 to $656 million for Scenario C2.

Morbidity. For the acute morbidity endpoints, the opportunity costs of illness (COI) approach is used to reveal avoided medical care expenses, reduced time spent in hospital visits, regained productivity from bed confinement, and reduced work-day losses of family members taking care of the sick persons. According to the World Bank (1997), the social cost of a hospital outpatient and emergency room visit is $23 per case in China. The cost of hospital admission for respiratory diseases is estimated to be $500 per case. An acute hospital visit is estimated to take an average of one day's sick leave. One case of hospital admission for respiratory diseases is estimated to average 14 days, with an additional three days of family member attendance. Here the potential averted expenditures associated with attempts to prevent disease, as well as the disutility on the affected individuals caused by air pollution, are omitted.

Viscusi et al. (1991) measured the trade-off rate for chronic bronchitis risk with the mortality risk of an automobile accident and suggested a median trade-off rate of 0.32. This implies that the potential WTP for reducing a case of chronic bronchitis is $19,200, $48,000, and $142,400 for the three VOSL scenarios. It is assumed that occurrence of chronic bronchitis lasts for 20 years on average. A social discount rate of 5% is applied to calculate the annual present value of avoided chronic bronchitis cases.

The benefits of avoided morbidity cases are estimated to range from $38 to $119 million under Scenario C1 and from $24 to $76 million under Scenario C2. The avoided work-day losses are estimated to be $13 million for C1 and $8 million for C2. In sum, therefore, the total health benefits of controlling emissions from the Power Scenario C1 are estimated to lie in the range of $190–1162 million (in 1998 price). The health benefits from the Industry Scenario C2 are estimated to be in the range of $121–741 million (in 1998 price). Table 6 summarizes the annual health benefit estimates of the two scenarios in comparison with the total annual costs. This analysis also shows that the mortality benefits account for the lion's share of the estimated health benefits, and the estimates are highly sensitive to the VOSL chosen. When the VOSL derived from the Chongqing WTP study ($150,000) is used, the health benefits of the C1 and C2 scenarios are $417 and $266 million, respectively.

The estimates of the health implications of air pollution improvements in this analysis are conservative. First, the study focuses on the investigation of acute mortality associated with air pollution and omits the long-term perspective of pollution exposure. Prospective cohort studies conducted in the US identified significant effects of fine particles and sulfate on mortality and morbidity at relatively low levels over long-term exposures (Dockery et al., 1993; Pope et al., 1995). Omission of the chronic effects of air pollution substantially underestimates the potential health benefits of pollution reduction in the long term.

Table 6
Annual health benefits and costs of the emission control scenarios (unit: US$ million per year in 1998 dollars)

Health benefits		Power scenario C1	Industry scenario C2
Mortality	Low	139	88
	Medium	347	221
	High	1030	656
Morbidity	Low	38	24
	Medium	57	36
	High	119	76
Work day losses		13	8
Total annual benefits (medium)		190–1162 (417)	121–741 (266)
Total annual costs		395	94

Second, this study only addresses the health implications of PM_{10} associated with fossil-fuel combustion. However, controlling emissions from power and industrial sources also induces ancillary benefits in the form of reduced emissions of a suite of co-emitted pollutants and their formation products (e.g. gaseous SO_2, NO_x, and ozone). Omission of these other pollutants leads to an under-estimation of the total health benefits of the control scenarios. Moreover, the estimates of cost-of-illness only reflect the present healthcare system in China. With healthcare reform and the growth of a market economy, it is estimated that healthcare in China will be more expensive in 2020. Therefore, the benefits of avoided illness could be potentially higher in future years.

4. Conclusions

Compared with the costs of the two emission control scenarios, the health benefits considerably outweigh the investments in pollution control in most cases (see Table 7). The health benefit/cost ratio of the Power Scenario (C1) is in the range of 1–5, and this ratio is in the range of 2–15 for the Industry Scenario (C2). In the medium case, the values are 2.0 and 5.4, respectively. The major uncertainty in the benefit/cost ratio lies in the economic value that can be placed on human health damage in China; undoubtedly, as China continues to develop in the future this value will rise dramatically. The results of the analysis show that controlling air pollution sources in Shanghai will induce significant health benefits. Controlling emissions from industrial sources is found to be more cost-effective to protect human health than controlling power-plant emissions in Shanghai. As learned from the previous study (Streets et al., 1999), this is due to the typical co-location of industrial facilities and residential areas. Nevertheless, the exposure assessment shows that despite significant reductions in pollutant emissions neither scenario brings the pollution concentrations to safe levels in all segments of the Shanghai metropolitan area. Therefore, more aggressive pollution control policies are needed to protect the health of Shanghai residents.

Shanghai represents a megacity in the developing world with growing prosperity. It faces both opportunities and challenges to control urban air pollution and develop a sustainable and harmonious future in the face of ever-increasing energy demand and population growth. The task of protecting the population from health deterioration and ecosystem damage requires a policy agenda that embraces a suite of measures across the whole economy, instead of one or two 'winning' policies. The potential of technological solutions for pollution abatement in Shanghai has been confirmed by Gielen and Chen (2001), and the potential of electricity demand management and other planning solutions has been demonstrated by Liu et al. (1997). Although costly, investments in cleaner technologies and pollution prevention and control will accrue major social and economic benefits for the city in the long term. Lastly, we argue that, in general, it is important in developing countries to include human health protection as a criterion when forming urban development policies and to make it an integral part of the public policy-making process.

In Shanghai, policymakers have recognized the adverse health impacts of air pollution through studies such as the one conducted by Chen et al. (2002). The conclusions and recommendations of that study on policy measures to effectively mitigate air pollution and human health impacts have been set as priorities in the 5-year planning process of Shanghai. However, more work is still needed to better understand the contribution of fossil-fuel use to fine particulate and ozone pollution and the development of effective mitigating policy measures. The findings in Shanghai also provide insight into the potential air pollution challenges for other large cities in China. Successful implementation of clean energy and air pollution control policy in the broader China will require improved rigor of supporting analyses, enhanced public awareness through outreach and education, effective pollution monitoring systems, and an open and inclusive decision-making process.

Table 7
Health benefit/cost ratios of the emission control scenarios

Case	Power scenario C1	Industry scenario C2
Low	0.9	2.5
Medium	2.0	5.4
High	5.6	15.1

Note: both the costs and benefits values are inflated to reflect prices in 2020.

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