Assessment of the Health Benefits of Controlling Air Pollution in Shanghai, China

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Abstract

The large urban centers of the industrializing countries of the world are experiencing severe air quality problems as their demands for energy increase faster than their ability to afford strong environmental protection. This situation is particularly true in the fast-growing part of Asia, where coal often provides the fuel for power generation and industrial development and where the transportation sector grows unchecked. This paper describes the development of an integrated assessment of urban air quality and mitigation options for the city of Shanghai, China. First, a sector-specific, gridded inventory of emissions of SO_2, NO_x, and particulate matter (PM) is developed. PM is divided into three size categories (TSP, PM_{10}, and $PM_{2.5}$) and split into carbonaceous and mineral classes. The URBAT model, a non-steady-state Lagrangian puff model, and related techniques are used to determine the spatial distribution of ambient concentrations of primary and secondary pollutant species.

Damage functions are developed to determine the effects of these levels on human health in the greater Shanghai area. Two control scenarios are developed (for power generation and industry), and their effects on emissions of each species are estimated. The health benefits of the control measures are determined, and their relative costs are calculated. The result is a rudimentary cost-benefit analysis that can be used by urban and environmental planners as a guide for determining the relative effectiveness of taking different courses of action.

1. Introduction

In recent years, many developing countries have enjoyed the fruits of industrial growth due to their vibrant economies. However, rapidly expanding megacities in developing countries have experienced a deterioration of air quality due to their heavy demands for energy, which has endangered the health of millions of inhabitants. For example, more than 250 million people presently live and work in cities that do not comply with World Health Organization guidelines for air quality. Most megacities in the world are in the developing countries, and many other cities in those countries are reaching the megacity status [1], particularly in Asia where fossil fuels (primarily coal) provide much of the energy for power generation and industrial development. Accordingly, emissions of SO_2, NO_x, and particulate matter (PM), and other species are expected to increase commensurately. Severe damage to natural and agricultural ecosystems is likely in the future without introduction of emission controls. Abatement technologies could cost US$10-30 billion annually, but still provide only limited protection.

Although the health and environmental impact is great, it is difficult for developing countries to invest in low-polluting technologies or emission-control measures because of higher capital investment costs and because technology investment requirements compete with other pressing needs. Accordingly, there is a need to prioritize to achieve the greatest benefits in such situations.

A damage function model that links energy-related emissions in urban areas to ambient concentrations, and subsequent population exposures and health damage, and compares the benefits of introducing advanced energy technology in an equivalent metric will be useful to country energy policy planners, as they weigh national funding priorities, and to international funding agencies who provide loans for energy and environmental investment projects.

Presently, there are many studies involving different disciplines that link the health impact of air pollution to economic impact from different perspectives for various environmental regulations and policy concerns. Levy et al. [2] have taken an integrated approach that translates emissions to local/regional ambient air concentration, then estimates the subsequent expected public health damage, and translates this to monetary values for two coal-fired power plants in the northeastern U.S. Delucchi et al. [3] analyzed the health and visibility cost of air pollution in terms of kilograms of motor vehicle emissions. Levy et al. [4] estimated the monetary relationship of public health to ozone reduction.

Advances in economics have resulted in meta-analyses of the value of life as well as the social costs of morbidity [5], [6], [7]. Economists have also started to develop geographic-specific tools that can reliably estimate the value placed on mortality and morbidity [8]. Recently, the U.S. EPA and U.S. DOE, together with several countries, including Korea, have begun studies to connect pollution reduction and monetary valuation. The establishment of a damage function has increasingly become critically important in comparing the cost of a pollution reduction technology to the benefits to public health and the environment.

In this paper, we investigated Shanghai, China, as part of the authors' continued efforts to integrate air pollutant modeling with health impacts and valuation [9], [10]. This megacity in China is positioned above all other cities in China in terms of living standards, per capita GDP, per capita energy consumption, and energy application technology [11]. Shanghai has already implemented many energy and environmental policies such as consumption control on coal and fuel sulfur content control. The experience of this important model city will inevitably influence the future of other major cities in China. By examining past and present air pollution problems and air quality management strategies, we hope that our study would provide a starting point of environmental policy consideration for megacities of the future.

2. Approach

The study area and emission inventories are described in this section. To assess air concentration levels, an air quality model is presented, along with the input parameters and meteorological data used in the study. Finally, the health impact methodologies are briefly discussed.

2.1 Study area

Shanghai is one of the most populous and prosperous urban centers in China. Situated on the estuary of the Yangtze River facing the Pacific Ocean, the Shanghai Municipality is home to more than 14 million residents, as shown in Figure 1. Its major neighbors are Jiangsu Province to the northwest and Zhejiang Province to the southwest. Power plants and major industrial facilities are located along the Yangtze River and Huangpu River, which runs through the downtown Shanghai and allows water access to the Yangtze River.

Energy use in Shanghai has traditionally been dominated by coal, accounting 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 [12], [13].

Recently, advancement in the economic condition has also nourished the development of environmental regulation and investments in environmental management and pollution controls. In the last ten years, environmental policies that include relocating heavy industries from the urban centers to less-populated remote areas, replacing residential coal use with natural 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. Ambient air quality monitoring data show that TSP and SO_2 concentrations in the urban area of Shanghai have dropped significantly, although current particulate levels in Shanghai are still high [14], [15]. Increasingly, NO_x emissions from motor vehicles are becoming problematic.

There is a transition in economic activities along with demographic patterns as more manufacturing businesses move out of the heart of the city. Focusing only on reductions in urban pollution may mask the risks of increased population exposures to air pollution in the city's outskirts. Another concern is the increased pollution in Shanghai from mobile sources owing to booming economic power.

2.2 Emissions estimation and disaggregation

One base year (1995) and one future year (2020) were selected to compare the policy options associated with a business-as-usual (denoted as "2020B" throughout the paper) and two emission control scenarios, one for power generation (denoted as "2020P") and another for industry (denoted as "2020I"). The power generation case consisted of building only advanced facilities such as coal gasification plants after the year 2010; in the industrial sector, we phased out all coal use from the central metropolitan area. These two control scenarios were costed out for comparison with health benefits due to reduced atmospheric emissions.

Emissions associated with energy use were estimated on the basis of the actual or standard emission rates by fuel and combustion technologies. In doing so, the RAINS-Asia model (Version 8) was used, which was designed to trace the causes and consequences of acid deposition on air quality in Asia [16], [17]. Emissions from non-energy activities were estimated from activity data contained in local and national statistical yearbooks. Emissions of particles from road dust were estimated using work by Ji et al. [18]. The distribution of small industrial facilities was taken from Zhao and Zhao [19], and we implicitly assume that this distribution did not change much prior to 1995.

Emission profiles for five major species were developed: SO_2, NO_x, TSP, PM_{10}, and $PM_{2.5}$, with particles being further divided into mineral and carbonaceous species, as presented in Table 1. The largest emitters were major point sources (power plants and industry), biomass burning, diesel vehicles, industrial processes such as cement manufacturing, and road dust.

These emissions were disaggregated at $0.1° \times 0.1°$ resolution over the gridded approximation of the municipal boundary (see Figure 1) using appropriate distributions of total and rural populations, small industrial facilities, road traffic, and river and oceanic shipping lanes. Thirteen power plants and five major industrial installations were treated as individual point sources, and placed at their latitude/longitude coordinates. Figure 2 shows illustrative emission profiles for SO_2 for 1995 and 2020B and carbonaceous PM_{10} for 1995 and 2020B. The relative size of the symbols for power plant and major industrial sources indicates the relative magnitude of the emission sources. It can be seen how emissions are clustered around the city center, the industrial complex on the Yangtze River, and smaller facilities along the Huangpu River. Figure 2 shows the carbonaceous PM_{10} profile for 2020, which reflects an emission decrease from power plants but an increase from rivers/sea lanes, in comparison with the 1995 case.

2.3 Air quality modeling

To estimate airborne concentrations and dry/wet deposition, the URBAT (URban-Branching Atmospheric Trajectory) model was used [20]. The URBAT model is derived from the ATMOS model [21], which was designed to simulate the transport and deposition of sulfur around megacities. The URBAT model is a three-layer Lagrangian puff transport model with puff branching, incorporating the first-order chemical conversion scheme and the first-order processes of dry/wet deposition. The model uses NCEP reanalysis data with 6-hour winds, mixing height, and precipitation for the year 2000 [22]. The model is source-oriented: every hour, a new puff is released at its source in an appropriate vertical layer, depending on the source type (surface or elevated) and release time of day (day or night). The model was tested for Beijing and Bombay [20] and subsequently was used to examine alternative SO_2 emission control scenarios for Shanghai, intended to demonstrate the benefits of adopting clean coal technologies [9].

The URBAT model simulations were conducted at a resolution of $0.1° \times 0.1°$ over the greater Shanghai area presented in Figure 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) was simulated separately. The components were 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 μm and 10 μm ($PM_{10-2.5}$) 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, were added together to obtain PM_{10} concentrations for each grid cell. The pollutant concentrations were calculated on a 1-hour basis; and the monthly, seasonal, and annual mean concentrations were derived for each grid. The particle size distributions by source type were consistent with the numerical modeling of particulate matter in Shanghai reported by Xu et al. [23]. Nitrate contributions to PM were not estimated at this time due to modeling difficulties.

2.4 Health impact and valuation methodologies

Because the power and industrial pollution sources are located in different places in Shanghai, a population-weighted air pollution concentration for total PM_{10} was imputed using the results of the URBAT model and the known population density in Shanghai across a $0.1° \times 0.1°$ grid for the two sectors. For health impact and valuation analysis, only total PM_{10} was examined because the health impact of PM_{10} was well known and has been previously characterized quantitatively. The health benefits associated with reduced ambient air concentrations of PM_{10} are determined from epidemiological studies from China whenever possible. In this analysis, health benefits as a result of having air pollution measures included the following: reduced mortality, hospital visits, emergency room visits, and chronic bronchitis.

Monetary valuation of these health benefits were then assigned using willingness-to-pay (WTP) and cost-of-illness (COI) studies from China specifically on air quality improvement. Modeled results from URBAT and health impact and valuation together provided the input parameters for the damage function, which can estimate the monetary values of health outcomes from changes in ambient PM_{10} concentrations. This then enabled us to compare the cost of pollution reduction against expected health benefits by different economic sectors.

3. Results and discussion

Results are presented primarily for the base year of 1995 and a future year of 2020. The 25-year time horizon was chosen to give a long-term view of pathways for energy and environment that dramatically illustrate the potential future consequences of present-day practices. Using the URBAT model, SO_2 and secondary PM concentrations, by particle size, were simulated for Shanghai as a combination of primary particles and secondary sulfate (at the present time nitrate is not included). The results include development of annual concentration profiles for urban and suburban areas. This was done for each emissions scenario.

Figure 3 shows examples of the annual-average SO_2 concentration profiles for 1995, 2020B, 2000P, and 2000I. The resultant concentration profiles show peak concentrations occurring over densely populated areas, with annual average concentrations exceeding the Chinese National Ambient Air Quality Standard for the Class II Area (60 $\mu g/m^3$) for most of the populated regions of the simulated domain. Annual-average SO_2 concentration for the base year of 1995 (Figure 3) was predicted to be about 57 $\mu g/m^3$, ranging from 15 to 158 $\mu g/m^3$ over the city and exceeding the standard around the urban center along the Huangpu River. For the period 1990-2000, measured annual SO_2 concentrations in Shanghai have a downward trend ranging from 45 to 105 $\mu g/m^3$ at the urban center and from 20 to 60 $\mu g/m^3$ city-wide [11]. Predicted SO_2 values are a little overestimated, but are comparable to monitored ones, which suggest that the current modeling produces reasonable results. For the 2020 business-as-usual case (2020B), predicted SO_2 concentrations range from 19 to 226 $\mu g/m^3$ with an average of 79 $\mu g/m^3$, and concentration maxima are predicted to occur along the Yangtze River, due to emission increases from power plants (see Figure 3). Under the power sector control scenario (2020P), concentrations range from 12 to 143 $\mu g/m^3$ with an average of 48 $\mu g/m^3$, while, under the industry sector control scenario (2020I), concentrations range from 16 to 211 $\mu g/m^3$ with an average of 66 $\mu g/m^3$. To compare concentration reduction benefits between scenarios, predicted SO_2 reductions for 2020P and 2020I with respect to the 2020B case were presented in Figure 4. The 2020P scenario would see its effects primarily along the Yangtze River where several large power plants are situated and to the south of the downtown area where several smaller power plants are located on the lower reaches of the Huangpu River. In contrast, the 2020I scenario would mostly affect the downtown area where the bulk of small, dispersed industrial coal combustion takes place. Note that population-weighted concentrations (taking the concentration at each 0.1º × 0.1º cell, multiplying by the affected population, and dividing by the total population) are about 20-30% higher than arithmetic averages over the city. This suggests that higher concentrations are concentrated on more populated areas.

General concentration levels and distribution patterns for PM_{10} are quite similar to those for SO_2, as shown in Figures 5 and 6. As was true for SO_2, measured annual TSP concentrations in Shanghai have shown a downward trend that ranges from 150 to 350 $\mu g/m^3$ for the period 1990-2000 [11]. Assuming a ratio of 0.6 for PM_{10} to TSP, measured PM_{10} concentrations can be interpreted as ranging from 90 to 210 $\mu g/m^3$. However, predicted annual PM_{10} concentrations ranged from 19 to 121 $\mu g/m^3$ with an average of 50 $\mu g/m^3$ for the base year of 1995 (see Figure 5). This underestimation can be attributable to pollution from outside the region into Shanghai and/or exclusion of a nitrate contribution.

Our case study focused on a health benefit comparison of reduced PM_{10} for industry and power sectors. We used existing epidemiological studies to establish dose-response coefficients for premature mortality and morbidity. Shanghai and regional-specific studies were used whenever possible, but due to the lack of available studies, health studies from U.S. and other countries were used as surrogates.

Particular matter of 10 μm and less has been demonstrated repeatedly to have significant association with premature mortality. Premature mortality is defined as death before the expected life expectancy. The results of a meta-analysis by Lvovsky et al. [24] involving the pooled weighted average from nine PM_{10} epidemiological studies are used. Mortality rate from this meta-analysis is reported to be 0.84% higher than baseline per 10 $\mu g/m^3$ increase in annual average PM_{10} concentration. For morbidity, our health benefit outcomes included emergency room visits, hospital outpatient visits, hospital admissions, and chronic bronchitis. Although no PM_{10} morbidity studies were available for Shanghai, a TSP study conducted in Beijing, China, found increased morbidity rates of 1.1% and 0.6% for every 10 $\mu g/m^3$ increase in TSP for hospital and emergency room visits, respectively. TSP was converted to PM_{10} using a PM_{10}/TSP conversion ratio of 0.6 [25]. Due to the lack of available data on the dose-response coefficient for hospital admissions in China, we adopted a U.S. hospital admission meta-analysis by Dockery and Pope [26]. Chronic bronchitis was used to indicate air pollution-induced chronic respiratory illness. The percent contribution to the health impacts of every 10 $\mu g/m^3$ increase in PM_{10} (dose-response coefficient) is presented in Table 2.

Annual health benefits from PM_{10} reduction were estimated by applying the dose-response coefficient to the population across the 0.1° × 0.1° grid. PM_{10} reduction is more effective in the power sector than industry for all health endpoints considered. The number of cases benefited per year for the power sector is approximately 1.5 higher than industry sector. This has to do with mostly with the locations of the power plants with respect to the population distribution. Figures 5 and 6 indicate that although industrial plants are located mostly in the urban area, because of the wide distribution of power plants in the Shanghai region, the total number of people benefiting from power sector reduction is greater. The total number of people is estimated at 193,000 for the power sector scenario compared with 123,000 from PM_{10} control in industry. Table 3 summarizes the number of people benefiting as a result of air pollution measures for the power and industrial sectors by health endpoints.

In order to make comparison with the cost of control measures, the next step was to place monetary values on mortality and morbidity figures discussed above. For mortality, we used the willingness-to-pay (WTP) approach to estimate the benefit associated with avoided mortality risk from improved air quality. 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 [27]. The study estimated a median value of a statistical life (VSL) at US$34,458. We used VSL of US$150,000 for Shanghai after adjusting for the marginal effect of income in Chongqing and the income differential in Shanghai.

For morbidity, the highest social costs per case are associated with chronic bronchitis (US$4,800), hospital admissions (US$500), emergency room visits (US$23), and hospital visits (US$23). We used a cost of illness approach that included not only money saved from medical expenses, but also opportunity costs such as productivity gained from having a better health condition and time saved from family visits at the hospital. The derivation of these values is provided in full in Li et al. [28].

The total health benefits per annum of introducing air pollution measures for the power sector is US$417 million (M) (lower bound: US$190M; upper bound: US$1,162M); while the benefit for the industry sector is US$266M (lower bound: US$121M; upper bound: US$741M) (see Table 4). Pollution control in either the power or industry sector will bring significant health benefits to the society. However, the benefit/cost ratio of 5.4 from industry vs. 2.0 from the power sector indicates that pollution control from industry outweighs that of the power sector [28]. This is due to the relatively high cost of reducing PM_{10} pollution in the power sector. Our integrated approach equates the cost of pollution reduction and the health benefit to one common metric so that the economic growth for Shanghai can be compared directly with the health benefits of pollution reduction.

4. Conclusion

Many megacities in developing countries in Asia are experiencing rapid growth of urban populations and unprecedented economic growth. The attendant energy demands associated with industrial/economic and population growth will be met by combustion of fossil fuels. However, improved control technology is presently unable to keep pace with economic growth. Accordingly, projections consistently show worsening air quality in many Asian megacities in the foreseeable future. Substantial increases in the use of indigenous coal as a primary fuel source account for much of the local air pollution. In order to make coherent policy choices by prioritizing investments to achieve the greatest benefits, it is necessary to quantify the health benefits associated with changes in potential emissions. This study demonstrates the potential benefits from emission controls and potential health damages associated with increased emissions. In doing so, we selected a base year (1995) and a future year (2020) with two (power and industry sector) control scenarios for Shanghai as a target city. Emissions were estimated on the basis of the RAINS-Asia database and available references and simulated using the URBAT model. Cost-benefit analysis associated with health impacts was exercised for total PM_{10}. One interesting finding from this study is that the power control scenario seems more beneficial in reducing the concentration levels, but the industrial sector control strategy seems to show the most favorable cost/benefit ratio. From these results, 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.

Compared with the costs of the two emission control scenarios, the health benefits considerably outweigh the investments in pollution control in most cases. The health benefit/cost ratio of the Power Scenario (2020P) is in the range of 1-5, while this ratio is in the range of 2-15 for the Industry Scenario (2020I). 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, because opportunity costs will also rise. 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 the human health than controlling power emissions in Shanghai. As was learned from the previous study [9], this is due to the typical co-location of industrial facilities and residential areas. However, the exposure assessment shows that despite significant reductions in pollutant emissions, neither scenario brings the pollution concentrations to safe levels in all parts of the Shanghai metropolitan area. Therefore, more aggressive pollution control policies are needed to protect the health of Shanghai residents, in the light of recent findings that there is no zero risk level for fine particle inhalation exposure of humans [29].

5. Acknowledgments

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6. References

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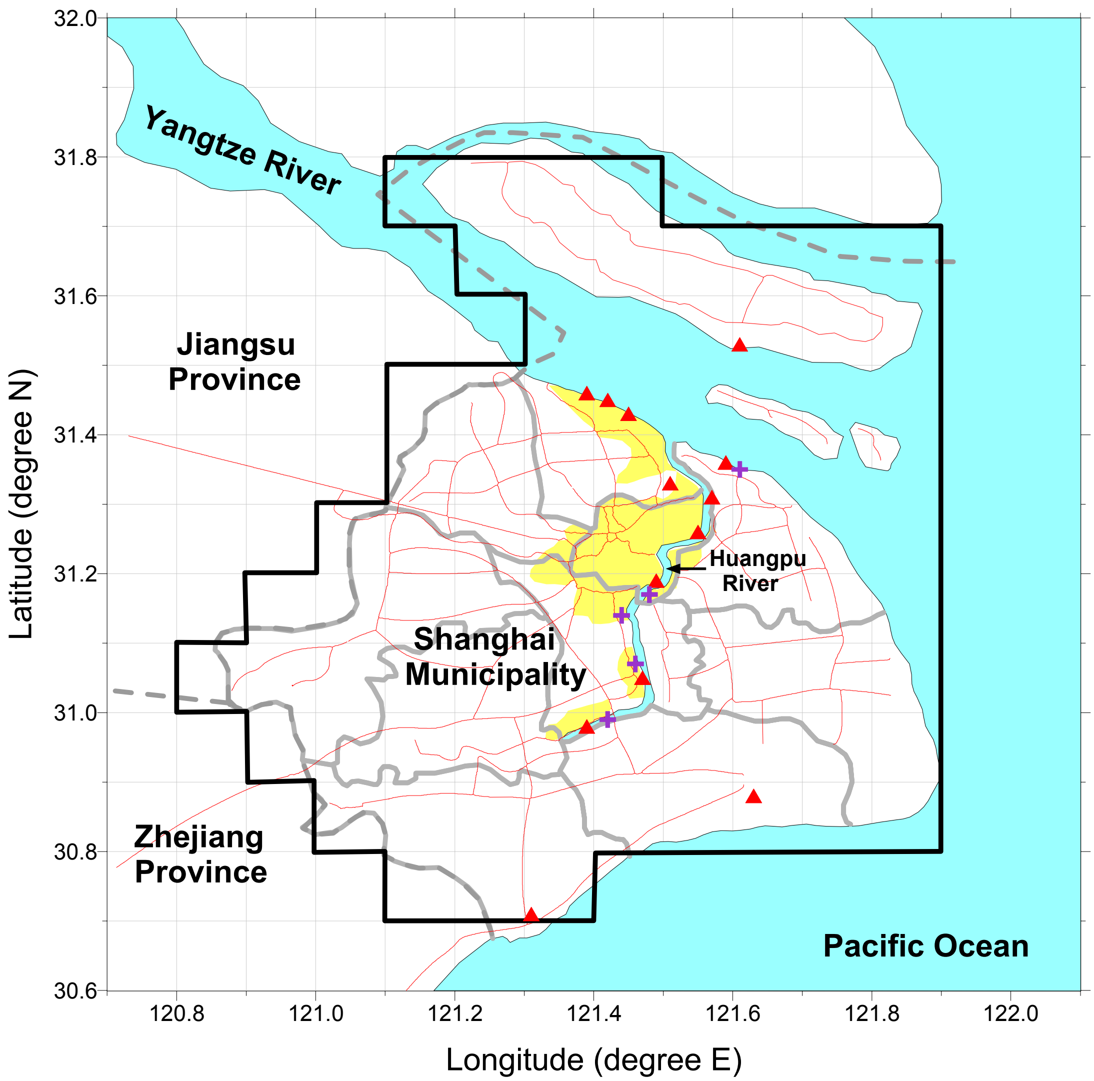

Figure 1. Map of Shanghai Municipality and modeling domain (solid line indicates 0.1° × 0.1° modeling domain; hatched area is Shanghai metropolitan area; triangles are power plants; and crosses are large industrial facilities)

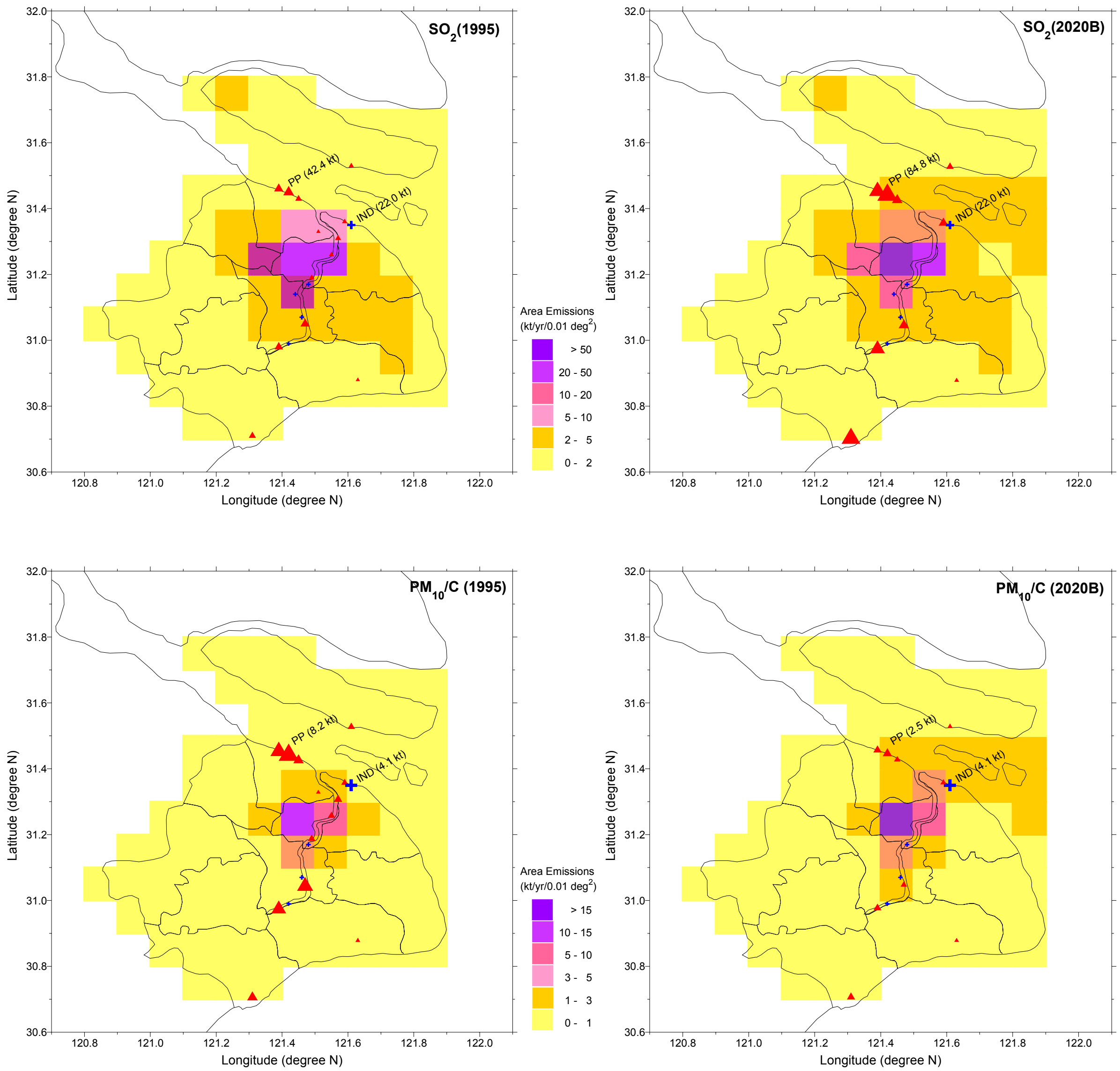

Figure 2. Estimated annual emissions distribution for SO_2 for 1995 and 2020 Business-As-Usual Case (2020B), PM_{10}/C for 1995 and 2020B

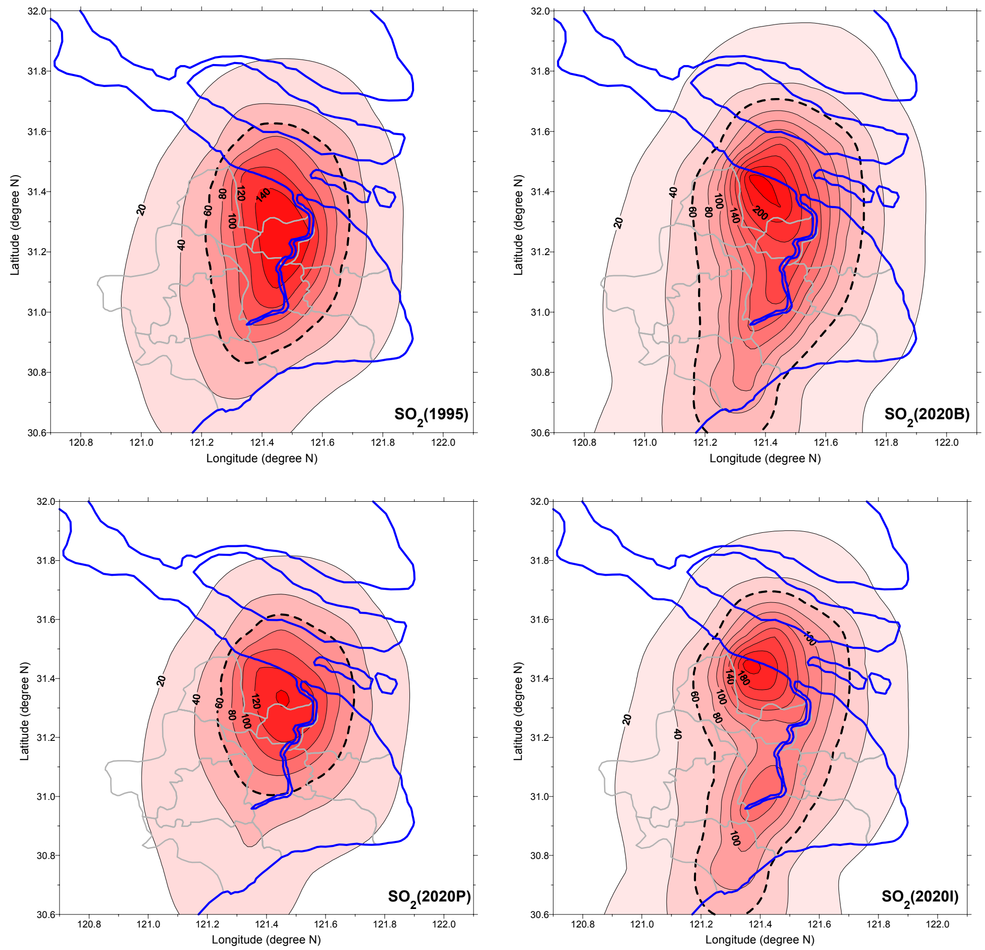

Figure 3. Predicted SO_2 concentrations ($\mu g/m^3$) for 1995, 2020 Business-As-Usual Case (2020B), Power Sector Control (2020P), and Industry Sector Control (2020I). Dotted line of 60 $\mu g/m^3$ denotes the Chinese National Ambient Air Quality Standard for the Class II Area.

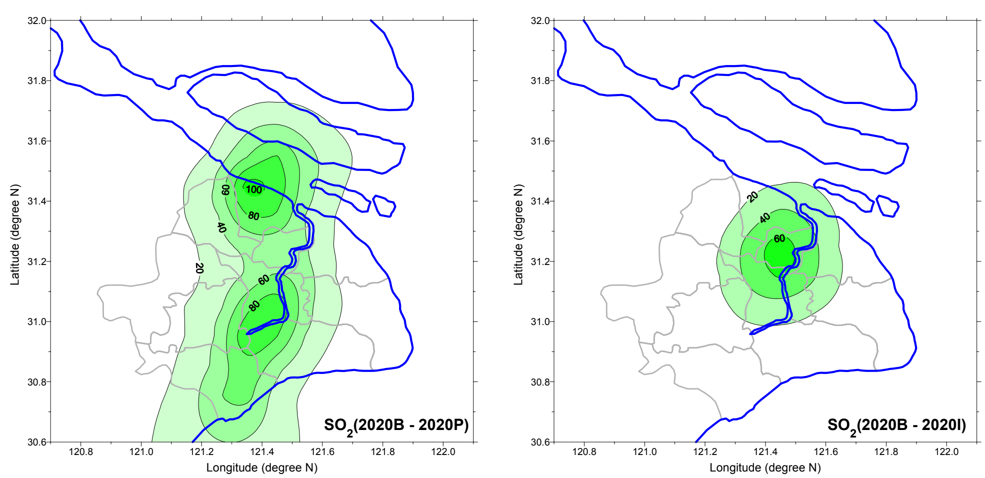

Figure 4. Predicted SO_2 reductions ($\mu g/m^3$) for Power Sector Control (2020B-2020P) and Industry Sector Control (2020B-2020I) with respect to the 2020 Business-As-Usual Case

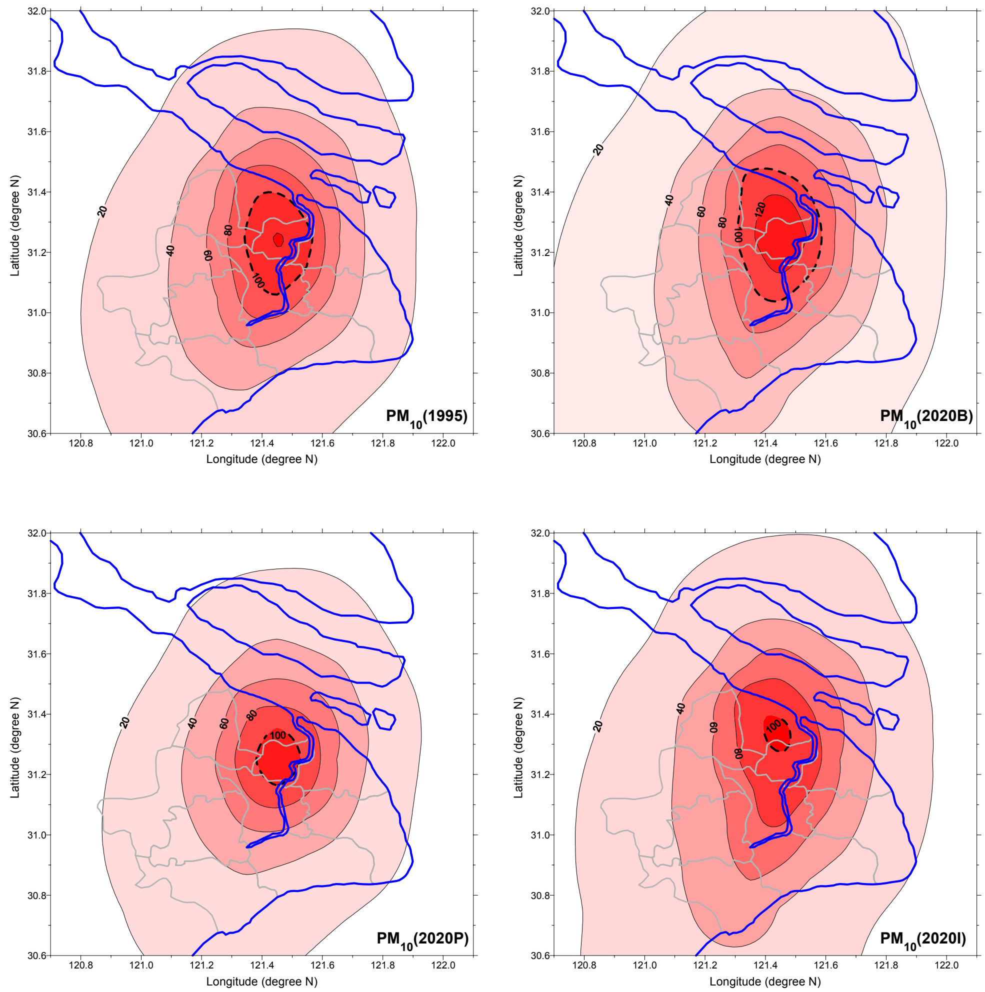

Figure 5. Predicted PM_{10} concentrations ($\mu g/m^3$) for 1995, 2020 Business-As-Usual Case (2020B), Power Sector Control (2020P), and Industry Sector Control (2020I). Dotted line of 100 $\mu g/m^3$ denotes the Chinese National Ambient Air Quality Standard for the Class II Area

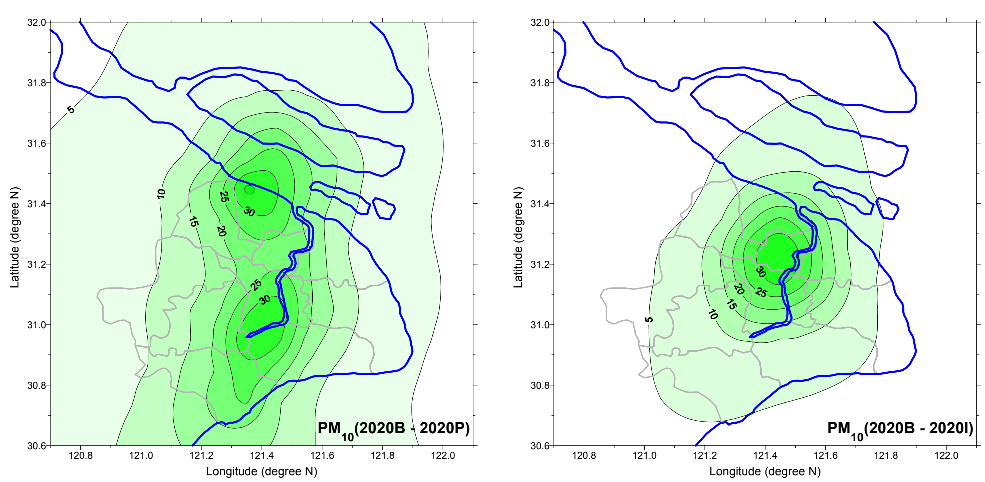

Figure 6. Predicted PM_{10} reductions ($\mu g/m^3$) for Power Sector Control (2020B-2020P) and Industry Sector Control (2020B-2020I) with respect to the 2020 Business-As-Usual Case

Table 1. Summary of Emission Estimations for 1995 (Base Year) and 2020 (Future Year) with Control Scenarios

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 source category with emission changes is presented.

Table 2. Central Estimates for Increase in Mortality and Morbidity Effect Per 10 $\mu g/m^3$ Increase in Annual Average PM_{10} Concentration

Health Endpoint	% Increase
Mortality	0.84 [a]
Hospital Outpatient 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)

Table 3. Annual Health Benefits from Pollution Control by Sector (cases/year)

	Pollution Control	
Health Endpoint	Power Sector	Industry
Mortality	2,808	1,790
Hospital Visit	96,293	61,379
Emergency Room Visit	48,506	30,918
Hospital Admission	43,428	27,716
Chronic Bronchitis	1,753	1,117

Table 4. Monetary Valuation from Pollution Control by Sector (US$ Million)

	Pollution Control	
Health Benefits	Power Sector	Industry
Mortality	347	221
Morbidity	57	36
Work Day Losses	13	8
Total Benefits	417	266