

URBANIZATION AND THE IMPACT ON HEALTH AND THE ENVIRONMENT: A TALE OF TWO CITIES*

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ABSTRACT

Rapid urbanization in China has increasingly impacted on the urban environment and the health of the growing urban population. Perhaps most visible is the smog that blankets the urban landscape. High ambient concentrations of airborne pollutants cause respiratory diseases among urban residents and also contribute to acid rain which damages the ecosystems beyond urban areas. In this paper, we use emission data collected in two Chinese cities to analyze pollution dispersion. Spatial models are developed to estimate population exposure to ambient concentrations of sulfate, a fine particulate most damaging to human health, and to assess the impact of acid deposition on the ecosystems surrounding urban areas. Dose-response functions are used to quantify the effects. The results show that the health costs are an order of magnitude larger than the damage to crops and forests. Policy implications are explored by evaluating the benefits and costs of alternative pollution control options.

Key words: *urban development, air pollution, environmental health, ecosystems, spatial modeling.*

JEL Classifications: O13, I100, Q010

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1. Introduction

Rapid urbanization in China is having an increasing impact on the health of the growing urban population (Liu et al. 2003). More than 450 million of China's 1.3 billion population are now living in urban areas where over 70 percent of China's gross domestic product (GDP) are generated, and where the environmental impacts of concentrated human activities are felt the most. Ensuring a livable urban environment is therefore a priority in China's urban development. An important dimension of a livable city is access to cleaner air, cleaner water, and basic environmental services for the urban residents.

Air pollution is one of the most visible environmental problems in China's urban life. High ambient concentrations of ground level emissions cast a hazy veil to the urban landscape, cause respiratory diseases among urban residents and also damage the ecosystems. A recent survey by the State Environmental Protection Administration (SEPA 2002) found that two out of three cities in China fail to meet the residential ambient air quality standard,¹ resulting in large population exposure to health risks such as chronic bronchitis, pulmonary heart disease, and lung cancer. In addition to the local health effects, air pollution has regional environmental consequences. Emissions from coal consumption by urban industries cause acid rain, which may be carried by wind to fall in adjacent areas, damaging water resources, forests, and crops.

Reflecting the spatial pattern of different pollution impacts, two control zones have been designated by China to target two concerns of urban air pollution: (i) the sulfur dioxide control zone and (ii) the acid rain control zone. The sulfur dioxide control zone covers 64 cities where the main concern is fine particulate matter emitted from coal combustion. The acid rain control zone encompasses 118 cities in regions across 12 provinces currently impacted by acid rain. Together the two zones cover 1.09 million square kilometers.

(Figure 15.1 about here)

This study selects two cities from the two control zones, respectively, to assess the health and environmental impacts of urban air pollution in China. In the sulfur dioxide control zone, the city chosen is Shijiazhuang in Hebei province where sulfur emissions have been deteriorating in recent years. Shijiazhuang is currently ranked the third worst air pollution city in China. In the acid rain control zone, the selected city is Changsha in Hunan province, which is one of China's grain basins but has been increasingly threatened by acid rain. The study goes beyond the city boundaries of Changsha to include the surrounding areas given the extent of ecosystem damage by acid rain in the region.

Spatial models are used to estimate ambient concentration and acid deposition for the two cities. We focus our analysis on emissions of sulfur dioxide (SO₂) from coal consumption, which accounts for about 70 percent of China's total energy use and is the major source of air pollution damage to human health and the environment in China. Despite improvement in recent years, more than half of the cities in the SO₂ control zone exceeded the annual concentration limit for SO₂ in year 2001 (SEPA 2002). SO₂ emission sources such as industrial boilers, furnaces, and stoves are a particular influence on urban air quality due to their low stack heights. In contrast, other major pollutants such as nitrogen dioxide (NO₂) from vehicle emissions are largely in compliance with air quality standards.²

The health effects of air pollutants are strongly linked to particle size. Scientific studies suggest that fine particulate matter – less than 10 microns in diameter (PM₁₀) – is likely to be most dangerous, because such fine particles can be inhaled deeply into the lungs where the clearance time of deposited particles is much longer, increasing the potential for adverse health effects. The most common airborne fine particles in China are

¹ China adopts three classes of ambient air quality standards. Class I are tourist, historic and conservation areas. Class II are residential urban and rural areas. Class III are industrial areas and heavy traffic areas. Chinese ambient air quality standards are comparable to those of the World Health Organization (WHO, 2000). For example, the Chinese standards for annual concentrations of sulfur dioxide are 20 and 60 micrograms per cubic meter (µg/m³) for Class I and Class II areas, respectively. The WHO guidelines for threshold values of annual average sulfur dioxide concentrations to protect public health are 40–60 µg/m³. For a detailed comparison, see Table 15.1.1 in World Bank (1997).

² Nitrogen dioxide (NO₂) from vehicle emissions contributes increasingly to urban air pollution but mainly in very large cities and the average NO₂ level is within the air quality limit (SEPA 2002). Lead (Pb) pollution has been dramatically reduced following the national ban on the production and sale of leaded gasoline on July 1, 2000. Carbon monoxide (CO) and ozone (O₃) are not systematically monitored.

sulfates, which are aerosols converted from SO₂ when condensed in the atmosphere, and are identified as the primary component of inhalable particulate matter and a chief culprit for health damage.

Emissions of SO₂ are also the precursors to acid rain. The sulfuric acids of SO₂ in the atmosphere react with humidity to produce acidic water droplets, which can be carried long distances by prevailing winds to fall as acid rain, snow, or fog. Such acid depositions cause soil and water acidification as well as surface material corrosion, resulting in losses of crops, forests, and fisheries, and building damage. In the acid rain control zone, the acidity of annual precipitations in 2001 fell below the safe pH threshold value of 5.6 in most cities (SEPA 2002).

This paper will use emission inventory data collected in the cities of Shijiazhuang and Changsha to develop pollution dispersion models and estimate ambient concentration and acid deposition.³ Dose-response functions will be used to quantify the effects of exposure to ambient concentrations on human mortality and morbidity, as well as the effects of acid deposition on crops and forests. A combination of willingness to pay and marginal social cost approaches are used in the economic valuation of these effects and alternative policy options.

2. Modeling Methodology

The analytical tools used in this study are the Urban Branching Atmospheric Trajectory (UrBAT) model and the Regional Air Pollution Information System for Asia (RAINS-Asia) model. UrBAT is a three dimensional multi-layered Lagrangian model capable of estimating ambient concentrations at urban scale (Calori and Carmichael 1999), while RAINS-Asia is a long range trajectory model developed to trace the long range causes and consequences of air pollution across 23 countries and 94 sub-regions in Asia (IIASA 2001; Shah et al. 2000). To capture the impact of air pollutants from distant sources, the resolution used in RAINS-Asia is typically 1° by 1° in latitude and longitude to accommodate analysis at a regional scale. UrBAT provides higher resolution analysis, using much of the same data. In this study, UrBAT is calibrated to the resolution of 0.01° by 0.01°, equivalent to approximately 1 km by 1 km.

(Figure 15.2 about here)

Figure 15.2 presents the schematics of the modeling framework. Meteorological fields, terrain features, population density, and emission sources are all incorporated as inputs into the UrBAT and RAINS-Asia models, to generate spatial patterns of pollution dispersion and population exposure. UrBAT and RAINS-Asia trace emissions from sources to receptors in the following manner:

$$R_i = \sum_j T_{ij} E_j$$

where R_i is a vector of ambient concentrations in area i , T_{ij} is a transfer matrix that determines the proportion of net emissions from area j transported to area i , and E_j is a vector of emissions from area j . Because of its fine resolution, UrBAT can thus relate emission sources to ambient concentrations within an urban area. In contrast, RAINS-Asia can trace the sources and consequences of air pollutant transportation and transformation between regions.

The UrBAT model separates the vertical dimension into two layers (mixing layer and upper layer) during the day and three layers (surface layer, boundary layer, and upper layer) at night as determined by the vertical temperature profile. The model allocates emissions in different layers according to the source type and the hour of the day. Area emission sources (small industries, residential and mobile sources) are released into the mixing layer during the day and into the surface layer at night. Elevated sources such as high stack emissions are also released into the mixing layer during the day, but at night they are branched into both the surface and boundary layers. In the RAINS-Asia model, elevated emissions are dispersed entirely within the boundary layer at night. Further details are documented in Arndt et al. (1998) and Calori and Carmichael (1999).

Emission plumes are modeled as puffs released in a given time interval, and the trajectory of each puff is tracked in the modeling domain. In this study, gaseous sulfur dioxide (SO₂) is assumed to emit from each

³ Two field missions to the cities of Shijiazhuang and Changsha were conducted in 2000 and 2001, sponsored by the World Bank. Data collections were made possible with valuable assistance from the State Environmental Protection Administration, the Chinese Research Academy of Environmental Sciences, and local Environmental Protection Bureaus in the cities of Shijiazhuang, Changsha, Xiangtan, and Zhuzhou.

source, and chemical conversion to aerosol sulfate (SO_4) follows a simple first-order rate constant, and rates of dry and wet deposition. Specifically, the mass of sulfur M and the partitioning of sulfur between SO_2 and SO_4 within each parcel j are calculated every hour by solving the following mass balance equations:

$$\begin{aligned}MSO_2(j, t+\Delta t) &= MSO_2(j, t)\text{Exp}(-\Phi \cdot \Delta t) \\MSO_4(j, t+\Delta t) &= K \cdot MSO_2(j, t+\Delta t)/(\Psi-\Phi) + \{[MSO_4(j, t+\Delta t)-K \cdot MSO_2(j, t+\Delta t)/(\Psi-\Phi)]\text{Exp}(-\Phi \cdot \Delta t)\}\end{aligned}$$

where t is the advection step time interval (seconds); $\Phi = K + V_{d-SO_2}/Z_i + k_{w-SO_2}$; $\Psi = V_{d-SO_4}/Z_i + k_{w-SO_4}$; K is the first-order chemical conversion of SO_2 to SO_4 (per second); V_d is the dry deposition velocity (meters per second) for SO_2 and SO_4 ; Z_i is the height of layer i (meters); k_w is the wet-removal rate constants (per second) for SO_2 and SO_4 .

The concentrations in a given layer, at each point within the model domain, are calculated based on the mass of each puff in the layer and are finally computed by superimposition of each individual puffs, as follows:

$$\begin{aligned}SO_2(lat, long, t+\Delta t) &= MSO_2(j, t+\Delta t) \text{Exp}(-r^2\sigma^2/2)/(2\pi\sigma^2LV) \\SO_4(lat, long, t+\Delta t) &= MSO_4(j, t+\Delta t) \text{Exp}(-r^2\sigma^2/2)/(2\pi\sigma^2LV)\end{aligned}$$

where σ is the horizontal mass standard deviation; L is the weighted layer depth; r is the distance from the puff center; and V is the volume of the grid centered at $(lat, long)$.

The meteorological data used to disperse emissions in different layers are obtained from the NCEP/NCAR Reanalysis Project (Kalnay et al. 1996). Information on wind vector, precipitation rate, mixing height, surface pressure, surface temperature, and surface heat sensitivity are extracted from this data source for the study domain.

Distant emissions from outside the urban area are modeled to form background concentrations. This is important as urban air quality is affected not only by emissions from within the city, but also by remote sources. In the case of Shijiazhuang city, there are a number of large point sources (thermal power plants, cement and lime factories) in the greater Shijiazhuang region upwind to the northwest of the city. Long-range transboundary emissions from neighboring provinces are considered by extracting information from the RAINS-Asia database (IIASA 2001). This is particularly useful for regional ecosystem analysis where the environmental impact goes beyond city boundaries such as the case for Changsha region.

Applications of the UrBAT and RAINS-Asia models to a number of cities and regions in China, including Beijing (Calori and Carmichael 1999), Shanghai (Streets et al. 1999), Chongqing (Adhikary 1988), and Jiangsu province (Chang 1998) suggest that UrBAT and RAINS-Asia are quite robust in their estimation for the purpose of human health and environmental impact assessment. In this study, the existing model parameters for UrBAT (Version 2.3) and RAINS-Asia (Version 7.52) are used unless otherwise cited.

3. Shijiazhuang

Shijiazhuang city, the capital of Hebei province, is home to more than 1.5 million urban residents in an area about 250 square kilometers. It is part of the Shijiazhuang “municipality,” an administrative unit comprising 17 counties and 222 towns in Hebei province. The municipality is 61 times larger in area but has only one-tenth the population density of the city. Serious environmental problems in the greater Shijiazhuang region include water quality and land degradation. This study, however, focuses on urban Shijiazhuang, where air pollution is the major environmental concern.

Shijiazhuang relies overwhelmingly on coal to power its urban economy. The share of coal is as high as 97 percent in total *primary* fuel consumption. Gas accounts for only two percent. Fuel oil makes up a mere one percent (processed fuels, such as petrol and diesel, are not included). To make matters worse, most of the coal consumed in Shijiazhuang has a rather high average sulfur content of 1.5 percent.

Within the city area, there are currently over 8100 coal-fired boilers and industrial kilns, among which 30 are large point sources with high stacks over 45 meters. Most of the latter are plants that supply electrical power and steam heat to industrial factories. The small sources, by contrast, are mostly in the industrial and residential sectors and have low emission heights. The large number of small area sources has important health implications as they contribute more to ground level concentrations than the elevated emissions from the power heat plants. The geographic conditions also exacerbate ambient pollution as the city lies at the foot of the Taihang Mountain, which reduces air dispersion above the city.

Data collected from Shijiazhuang show that the annual average SO₂ concentration was 130 micrograms per cubic meter (µg/m³) in 1999, more than twice the limit of 60 µg/m³. Violations were the worst in winter months, reflecting both increased coal use for heating and reduced air dispersion in winter due to meteorological conditions.⁴ As a result, Shijiazhuang suffers serious environmental health consequences. Available health statistics suggest that about four percent of the city's residents suffer from chronic bronchitis, pulmonary heart disease, and lung cancer. Li and Ren et al. (1998) reported that the incidence rates were higher in "polluted" areas than in "clean" areas, and the differentials of the incidence rates were 9 cases per thousand residents for chronic bronchitis, 11 per thousand for pulmonary heart disease, and 8.33 per a hundred thousand for lung cancer.

Detailed energy consumption and emission data for 2000 were collected which enabled tracking energy emissions to their sources by location. Based on emission patterns in Shijiazhuang, the main sources of air pollution are fine particulate matter from coal combustion such as soot, flying ashes, and especially sulfate. Contributions from other sources such as road and construction dust are small.

A matrix was developed which divides the Shijiazhuang urban area into 1 km by 1 km grid cells, with 19 east-west grids and 13 north-south grids. The spatial distribution of SO₂ emissions in Shijiazhuang in 2000 is presented in Figure 15.3. Over 55000 tons of SO₂ were emitted in Shijiazhuang city in 2000. One-third of the emissions were from space-heating boilers scattered in the residential areas, and about 15 percent from small and medium-sized industrial boilers and kilns. The rest are attributed to the large power-and-heat plants.

(Figure 15.3 about here)

In addition to emissions from the city area, distant emissions from outside the Shijiazhuang urban area are also estimated. In 2000, the large thermal power plants and cement factories upwind of the city emitted 61000 tons of SO₂, more than those emitted from within the city area. These emissions are identified by location in the greater Shijiazhuang region and are incorporated in the UrBAT model to estimate their impact on urban air quality in Shijiazhuang city. Also, the long-range transboundary emissions from outside the Hebei province are extracted from the RAINS-Asia database (IIASA 2001) to generate concentrations for the urban area.

Figure 15.4 shows the annual average SO₂ concentrations in 2000 as estimated by UrBAT, incorporating the emissions from all sources. Large areas of urban Shijiazhuang are found to have SO₂ concentrations far above the annual average limit of 60 µg/m³, with the northeast area of the city over twice this value. These results are consistent with monitored SO₂ levels. For example, the annual average SO₂ concentration calculated from air samples collected within the second ring road surrounding the city was 112 µg/m³ for 2000 (Shijiazhuang Municipal EPB 2001). The ring roads are shown in Figure 15.4 to delineate the layout of the Shijiazhuang urban area but are not related to the pattern of SO₂ concentrations.

(Figure 15.4 about here)

The concentrations of sulfate are presented in Figure 15.5. The impact of outside sources upwind to the northwest of the city is reflected in the west suburbs, but inside sources from the northeast area of the city dominate the sulfate concentration pattern. No guidelines have been set specifically for ambient sulfate concentrations. But evidence from a study by Dockery et al. (1993) shows that for fine particulate matter (PM₁₀), there is a significant relationship between mortality rates and PM₁₀ exposure, with a demonstrated 0.8 percent increase in mortality for each 10 µg/m³ annual increase in PM₁₀ exposure.

(Figure 15.5 about here)

The spatial pattern of ambient concentrations shown in Figure 15.4 and Figure 15.5 are superimposed on the population profile of Shijiazhuang to estimate population exposure. We use the LandScan 2000 global population database (Oak Ridge National Laboratory 2001), which apportions sub-national census data based on the locations of roads, slopes, land cover and nighttime lights in order to develop a population matrix for Shijiazhuang in 1 km by 1 km resolution (Figure 15.6). It is estimated that, among the 1.53 million residents in the urban areas of Shijiazhuang, about 1.4 million (92 percent) were exposed to various degrees of

⁴ Shijiazhuang's annual average NO_x concentration in 1999 was 75 µg/m³, within the standard of 100 µg/m³. NO_x pollution has been increasing in recent years due to an increase in the number of motor vehicles.

health-damaging SO_2 and SO_4 concentrations above the threshold levels of $60 \mu\text{g}/\text{m}^3$ and $10 \mu\text{g}/\text{m}^3$, respectively, in 2000.

(Figure 15.6 about here)

4. Changsha

The Changsha region in Hunan province is one of China's grain basins, blessed with fertile soil and favorable subtropical monsoon climate for crops. In recent years, this grain basin has been increasingly threatened by acid rain. Large areas of vegetation turned yellow following acid rain. Paddy, the main grain grown locally has suffered acute injury and reduced yields in many areas of the Changsha region. Crop losses claimed by farmers are in the amount of millions of RMB.⁵

Together with two nearby cities, Changsha city forms a triangle with Xiangtan city at the lower left tip and Zhuzhou city at the lower right tip, all within the reach of 50 kilometers from one another. The entire triangle area is impacted by acid rain, with the average value of pH at about 4.2 for annual precipitation in the region (Figure 15.7). Some rainfalls have pH levels lower than 3, far below the 5.6 safe pH threshold value. This is especially damaging for the region because the local soil type is red soil, which is not assimilative to acid compounds.

(Figure 15.7 about here)

Different from Shijiazhuang where the northern dry climate makes airborne sulfur mainly a respiratory hazard, sulfur emissions in the Changsha region do the damage largely in the form of wet deposition due to high humidity in the region. As sulfuric acids are easily dissolved in water, sulfur emissions react with the high humidity in the atmosphere to produce acid rain.

Similar to the cause of air pollution in Shijiazhuang, the chief culprit for acid rain in the Changsha triangle is coal, which accounts for over 80 percent of total energy consumption in the region. Most coal is produced locally, with sulfur content over three percent, which more than doubles that of 1.5 percent in Shijiazhuang. Some coal in the Changsha region even has sulfur content as high as six percent. The resulting emissions from coal consumption are high in sulfur.

Over 200 kilotons of SO_2 were emitted in the Changsha triangle in 2000. The sources of SO_2 emissions in Changsha are mainly small and medium-sized coal boilers typically with stack height lower than 50 meters. The spatial density of emission sources is about 60 boilers per square kilometer. Helped by the geographic conditions that the entire region is located in the valley of the Xiangjiang River, the surrounding mountain ranges act to reduce airflows and trap the sulfuric acids in the atmosphere, the acid rain brewed by local sulfur emissions falls mostly within the region.

Results of computer analysis of the rain samples collected in Changsha are presented in Table 15.1. Twelve chemical indexes are measured in addition to pH. Lab analysis of the data series shows that the main negative ions in the rain samples were SO_4^{2-} and NO_3^- , and the main positive ions were Ca^{2+} and NH_4^+ . The ratio of $(\text{SO}_4^{2-} + \text{NO}_3^-)/(\text{Ca}^{2+} + \text{NH}_4^+)$ was in the range of 0.86~1.49, indicating that SO_4^{2-} and NO_3^- were more active ingredients responsible for the acidity in precipitation. Between SO_4^{2-} and NO_3^- , the ratio ranged 5.38~11.81 in favor of SO_4^{2-} , pointing to sulfur as the key factor in the formation of acid rain in Changsha. However, the ratio of SO_4^{2-} to NO_3^- has been falling in recent years, which coincides with an increase of NO_x concentrations in air quality monitoring data, suggesting NO_x is an increasingly important pollutant in Changsha.

(Table 15.1 about here)

The environmental impact of sulfur emissions in the Changsha region is assessed by using the RAINS-Asia model, which takes socioeconomic and energy consumption data as input and incorporates

⁵ RMB stands for *Ren Min Bi*, Chinese currency. Data collected in Changsha region indicate that, since 1995, the Zhuzhou Smelter alone has paid RMB 1.5~2.5 million each year to the surrounding farming villages for grain loss compensation. Cases of farmers suing industries for crop damage by sulfur emissions from nearby factors take place frequently in recent years in the Changsha region. Local Environmental Protection Bureaus often act as mediators to resolve the cases by estimating crop losses based on historical yields.

sufficient atmospheric and technical details to generate estimates of emissions and agro-ecosystem damages. Figure 15.8 shows that the acid deposition originated from the Changsha triangle largely fell in the greater Changsha region where annual acid deposition is estimated to be between 1000~2000 $\mu\text{g}/\text{m}^2$, but other regions in Hunan and neighboring provinces are also affected by receiving 100~500 $\mu\text{g}/\text{m}^2$ of acid deposition in 2000.

(Figure 15.8 about here)

5. Cost of Sulfur Pollution

The analysis of Shijiazhuang and Changsha highlights the damage of sulfur pollution to both human health and the environment. To what extent they compare to each other? A quantification of the air pollution cost will help us gain an insight in the magnitude of the problem and also provide a basis for cost-benefit analysis in identifying optimal pollution control strategies.

Extensive clinical, epidemiological and toxicological studies provide evidence of the relationships between exposure to ambient concentrations and human health. Correlation among concentrations of different air pollutants, however, makes it hard to separate the effect of any single pollutant species on health. Data compiled from the Chinese epidemiological studies suggest that, of the many air pollutants, SO_2 and TSP cause the most extensive damage.⁶

While both SO_2 and TSP are found to be statistically significant in association with health outcomes, only TSP is measured consistently in China, and the most significant TSP associated with health problems is PM_{10} , which makes about 60 percent of TSP in China. PM_{10} is therefore adopted as the key measure for estimating health impact in this study, and sulfate is used as the proxy for PM_{10} . Using dose-response functions from both Chinese and international studies (the latter supplements those endpoints that were not measured directly in China), the health impact of high ambient concentrations in Shijiazhuang are estimated following the analytical method outlined by Ostro (1996):

$$dH_i = r_i \cdot \text{POP}_i \cdot (dA)$$

where dH_i is change in population risk of health effect i ; r_i is rate of change in dose-response function i ; POP_i is population exposure to health risk i ; dA is change in air pollution.

The economic valuation of health effects is based on estimates of willingness to pay to reduce risks of injury or death. Based on a survey conducted in Chongqing in 1998 by Chinese researchers using the risk-dollar trade-off method to reveal the implied value of a statistical life (Li and Schwartz et al. 1998), the median value of willingness to pay in a sample of 500 for avoiding a death was US\$160,000.⁷

Table 15.2 presents the resulting estimates of health costs of sulfate concentrations in Shijiazhuang, counting only those concentrations above the national ambient standard. Based on the population exposure to health-damaging SO_4 concentrations above 10 $\mu\text{g}/\text{m}^3$ as generated by UrBAT, it is estimated that 251 premature deaths, 7.7 million cases of acute and chronic morbidity, and 6,589 person-years of restricted activities would have been avoided if Shijiazhuang had met the national ambient air quality standards in 2000. Applying the economic valuation numbers, this implies US\$40 million for premature deaths and US\$31 million for morbidity. The total health cost of US\$71 million is about 4.3 percent of Shijiazhuang's GDP in 2000.

⁶ Xu and Gao *et al.* (1994) used a Poisson linear method to regress daily mortality and morbidity against the logarithm of SO_2 and TSP in Beijing, and found that the risks of chronic obstructive pulmonary disease increase by 29 percent and 38 percent with a doubling in SO_2 and TSP concentrations, respectively, and the risks of total mortality increase by 11 percent with a doubling in SO_2 concentrations. In other studies focused on TSP in Beijing (Xu, Li and Huang 1995; Xu and Wang, 1993), a 100 $\mu\text{g}/\text{m}^3$ increase in TSP concentrations is associated with 1.1 percent more hospital outpatients and 0.6 percent more emergency room visits. Similar studies conducted in other Chinese cities such as Shanghai (Chen, 1994), Shenyang (Xu and Yu *et al.* 2000), and Chongqing (World Bank, 1996) all find significant correlation between ambient concentrations and respiratory diseases and premature mortality.

⁷ Converted from RMB using the 1998 exchange rate of 8.28 RMB/US\$. The survey was conducted in RMB. Annual wages in 1998 were RMB 7479 for China on average and RMB 6433 for Chongqing, equivalent to US\$ 903 and US\$ 777, respectively.

(Table 15.2 about here)

These results are in line with the health cost estimates by local researchers in Shijiazhuang (Li and Ren et al. 1998). Using incidence rates and survey data, Li and Ren et al. (1998) estimated the costs of public health associated with air pollution in Shijiazhuang to be RMB 523 million in 1995, equivalent to US\$63 million converted at the 1995 exchange rate of 8.35 RMB/US\$. The higher value estimated for 2000 in this paper may reflect the urban population growth that increases population exposure, and also a 25 percent rise in monitored TSP concentrations (Shijiazhuang Municipal EPB 2001). However, our estimate is smaller as a share of GDP than that of Li and Ren et al. (1998) because of Shijiazhuang's rapid real GDP growth in recent years (13.2 percent per year from 1995 and 2000, according to Shijiazhuang Municipal EPB 2001).

The methodology used in this study to assess the health effects in Shijiazhuang has been applied to a number of other cities in China, including Beijing (Calori and Carmichael 1999), Shanghai (Streets et al. 1999), and Chongqing (Adhikary 1988). The estimates provided in these studies were generated on a consistent basis, which enables a ballpark extrapolation of the health effects of urban sulfur pollution in China. Using the same standard of 10 $\mu\text{g}/\text{m}^3$ for sulfate concentrations, Streets et al. (1999) projected the population exposure to unsafe sulfate concentration in Shanghai to be 2.5 million for year 2000 in a business as usual scenario, while Adhikary (1998) estimated the human exposure to sulfate concentration risks in Chongqing to be as high as 17 million for the same scenario. Assuming a linear extrapolation, the UrBAT estimates suggest that an average of 250 million urban residents in China may be exposed to health-damaging ambient sulfate concentrations in 2000.⁸ Using the same dose-response coefficients as in Table 15.2, the total health costs associated with China's urban air pollution would be more than US\$34 billion.

Comparing the health effect to ecological impact, a similar dose-response function approach can be used to assess the ecosystem cost. The exact impact of acid rain depends on the carrying capacity of soils and surface waters. The RAINS-Asia model incorporates critical loads for a variety of ecosystems to determine the assimilative level of acid compounds for different ecosystems in different regions. Figure 15.9 shows the excess acid deposition above critical load for the ecosystems in Changsha and other regions for a business as usual scenario. It is found that acid rain threatens large sensitive areas, especially in the southwestern region around Chongqing, the south central region around Changsha, and the east coast near Shanghai. It is estimated that nearly 30 percent of the total land area in China is exposed to acidification of soil and water above the tolerant capacity of the ecosystems.

(Figure 15.9 about here)

A study conducted in Chongqing provides information to arrive at a stylized relationship between acid deposition and crop and forest loss.⁹ Applying the excess acid deposition data generated by RAINS-Asia to an exponential dose-response function that relates annual sulfur deposition to crop/forest losses (World Bank 1997, Table 15.1 in Annex 2.2), the crop and forest losses alone amount to over RMB 1 billion in Hunan province, about 2 percent of the province's GDP. Comparing to the earlier estimate of health cost in terms of GDP, this is about half the size of the 4.3 percent GDP loss in health damage for Shijiazhuang. Assessment for all the acid rain impacted areas in China gives a total cost of over US\$4 billion for crop and forest losses in 2000.

The cost estimates for acid rain are limited only to crop and forest losses. Other damages to surface waters (fish and other aquatic species), surface materials (buildings and bridges), as well as to tourism, have not been accurately calculated in the literature. Perhaps more importantly, the damage of acid rain to environmental resources directly threatens the livelihood of many farmers, especially the poor who supplement their inadequate income by obtaining a substantial part of their consumption from natural and community resources.

⁸ China's latest census (Population Census Office, 2001) estimates the total urban population in 2000 to be over 450 million, and the population in Shanghai and Chongqing in 2000 to be 17 million and 31 million, respectively. Based on the population exposures projected by UrBAT, proportions of urban residents exposed to unsafe sulfate concentrations range from 15 percent in Shanghai, 55 percent in Chongqing, and 92 percent in Shijiazhuang.

⁹ In 1995, the Chongqing Environmental Protection Bureau conducted the most detailed analysis to date on crop and forest loss due to acid rain. Surveys in nine central districts of the Chongqing Municipality found that about 24 percent of the vegetable crops were damaged by acid rain in 1993, amounting to a loss of about RMB 62 million. Similar losses were found for grain crops, totaling about RMB 184 million, and for forests (both trees that died and for reduced growth) at RMB 169 million (World Bank, 1997).

They cultivate, fish, and graze livestock. Environmental resources play a vital role in their livelihood strategies because most of their incomes are obtained from ecological resources (Parikh 1998). When environmental resources degrade, their poverty becomes more acute as their options to take defensive actions against adversity are reduced. These social welfare implications on farmers are not reflected in the acid rain cost estimates.

6. Assessment of Control Options

In light of the identified costs associated with urban sulfur emissions, policy options are assessed here by cost-benefit comparisons where the benefits are measured in terms of the *avoided* damage to health and the ecosystems as estimated above but under alternative policy scenarios. Such cost-benefit comparisons evaluate the impacts associated with emission reductions. This is in contrast to the Total Emission Control Plan currently enforced top-down from the central government in China by assigning emission quotas to cities and local administrative areas, which focuses exclusively on the volume of emission reductions, but the resulting benefit may not be the largest. For example, large point sources such as power plants account for a bulk of SO₂ emissions, but contribute far less to ground-level concentrations due to elevated emission height. Cutting emissions from large point sources may help reaching an emission reduction target, but may not be effective in alleviating ground-level ambient concentrations that harms human health, which is most costly among all sources of damage by sulfur pollution.

Actions taken by Shijiazhuang and Changsha to reduce emissions reflect specific local conditions, but their measures fall largely in the following four categories:

(1) Low sulfur coal

Coal dominates energy use in both Shijiazhuang and Changsha. The potential to reduce SO₂ emissions by using low sulfur coal is enormous. Both cities have put in place regulations to restrict the sulfur content at 1 percent for all coal used in city areas. The average sulfur content of the coal used in Shijiazhuang is 1.5 percent. Changsha doubles that at 3 percent. A sulfur content limit of 1 percent would reduce sulfur emissions by a third in Shijiazhuang, and two thirds in Changsha.

(2) Small sources abatement

The overwhelming source of SO₂ emissions in both Shijiazhuang and Changsha is from small area sources. A drastic measure adopted by both cities has been to shut down small boilers and replace them with larger and more efficient facilities that reduce both energy consumption and emission. This has taken place mainly in the space heating sector. In Shijiazhuang, the coverage of central heating now reaches half of the city area. Similarly, central heating plants that are traditionally not built in southern regions such as Changsha are now planned to replace small heating boilers over the next 10 years.

(3) Energy conservation

China has made significant progress in energy conservation in the past two decades by holding the energy elasticity at 0.5 or below, meaning that energy consumption is reduced by half but still sustains the growth of the economy. Despite such unprecedented success, there appears to be scope for further improvement. This is no exception in Shijiazhuang or Changsha. A large number of the industrial plants in these two cities are heavy industry type, such as metal smelters and chemical factories, built in 1950s and 1960s. The efficiency of the technology used in these plants is far below current standards. Equipment upgrading and a structural adjustment toward less energy-intensive industries will considerably reduce energy consumption and emission. The potential of energy conservation is projected to be a continuing decline in energy intensity at 4.5 percent annually in the coming years.

(4) Fuel substitution

Diversification to non-coal sources is actively explored by Shijiazhuang and Changsha in an effort to increase the use of cleaner fuels, especially natural gas. Only a fraction of the energy currently used in both cities is gas, and is produced costly in the form of coal gasification or liquefied petroleum gas. However, two pipelines being built in China to transmit natural gas from the remote northwest to the east region will change the

landscape of energy supply considerably in areas along the pipelines. Tapping the Shaanxi-to-Beijing pipeline, Shijiazhuang is projected to reduce its reliance on coal by about 10 percent. Similarly, Changsha will access the Tarim-to-Shanghai pipeline to gasify its residential and commercial applications and some industrial production.

Assuming all four control options explored by Shijiazhuang and Changsha are implemented in a timeframe by year 2020 in all cities in China, the resulting benefits and costs are compared in Table 15.3 based on estimates generated by UrBAT and RAINS-Asia. The health benefits are estimated as the avoided costs associated with reductions in mortality and morbidity in year 2020 resulting from reduced population exposures following emission abatement specified by the control options. The crop/forest benefits are derived by extrapolating the damage estimates for 2000 to 2020 at a constant US\$200 per ton of SO₂ (World Bank 1997, p. 27). The avoided costs of crop/forest damage are then calculated in line with the emission reductions associated with each control option.

(Table 15.3 about here)

The results show that, among the four options, the low sulfur coal ranks the highest in terms of both health benefits and crop/forest savings, worth US\$54.3 billion and US\$3.4 billion, respectively. The potential of sulfur reduction by using cleaner coal is enormous. This is especially attractive because cities in China are only likely to reduce their dependence on coal gradually.

However, the regulatory approach to mandate a 1 percent sulfur limit, as easy as it may be simulated in a model, is difficult to monitor and enforce in an increasingly market-oriented Chinese economy. A major barrier is the lack of economic incentives to use low sulfur coal, which costs RMB 20~30 more per ton than the average coal. Coal sample tests done by the Shijiazhuang EPB in 2000 indicated that only about half of the coal used in Shijiazhuang was under the 1 percent sulfur content limit, despite Shijiazhuang's proximity to Shanxi where low sulfur coal is produced. For Changsha, given the high-sulfur coal produced locally, a tax outweighing the transport cost to bring low sulfur coal to the Changsha region is justified. Market-based incentives such as a pollution tax on dirty coal will be the key to support the low sulfur coal initiative and encourage coal users to switch to cleaner coal.

Consistent among all four options, the benefits derived from health (column 1) are an order of magnitude larger than those from crops and forests (column 2). This highlights the importance of health in assessing the impacts of emission reductions. The significant health benefits reflect future rapid growth of urban population and the associated mortality and morbidity otherwise caused by urban air pollution if uncontrolled. In the meantime, while the crop/forest benefits appear to be small next to the health estimates, the synergy of the four options in protecting both human health and crop/forest is in effect substantial considering the baseline estimate of the crop/forest damage was US\$4 billion for year 2000.

All four options involve substantial abatement costs as estimated by UrBAT and RAINS-Asia and also based on parameters derived from a number of studies (Shah et al. 2000). Dividing the benefits by abatement costs, the benefit/cost ratios in Table 15.3 show that the most efficient option is energy conservation. Indeed, the potentials of energy savings can be tapped through cost-free policy reforms such as removing market distortion of energy subsidy to state-owned enterprises. This will increase energy efficiency and reduce emissions, as well as improve the competitiveness of enterprises. Also cost effective is simply to provide information to energy users on energy saving opportunities in household appliances, building designs, transport and industrial motors. This complements the new *Energy Conservation Law* as implemented intensively in areas such as of minimum energy efficiency standards, monitoring energy use in industry, energy certification for products, and energy labeling.

Another insight from the benefit/cost comparisons is that of an integrated approach to pollution control. By targeting acid rain alone, three out of the four options (except energy conservation) seem to be not justified on a benefit/cost basis. The scope of fuel substitution into natural gas appears to be particularly limited. But energy conservation by itself is unlikely to be sufficient to prevent further large increase in sulfur emissions. It is important to adopt a comprehensive strategy that encourage energy saving, promote clean fuels such as low sulfur coal and natural gas, and also target small area sources to reduce emissions in a most cost effective manner. Large point sources such as power plants, which are largely uncontrolled in Shijiazhuang and Changsha, also need to be controlled. These options are not mutually exclusive and would need to be integrated, based on technical, social and political constraints in local cities.

7. Conclusions

As urbanization continues and real incomes grow in China, health and environmental concerns have increasingly impacted on the value and welfare of the urban population. Sulfate concentrations, which are fine particulates most damaging to human health, are highest in large urban areas and industrial centers such as Shanghai, Shenyang, and Shijiazhuang. Beyond urban areas, emissions from urban consumption of fossil fuels, coal in particular, result in acid rain that damages the ecosystems in many regions, especially in Chongqing, Guizhou and Changsha. This study selects two cities, Shijiazhuang and Changsha, to assess the health and environmental effects of urban emissions.

Using an atmospheric dispersion modeling approach, spatial models are developed to identify the sources of emissions to estimate population exposure to high ambient concentrations of sulfate, and to assess the acid deposition exceeding the critical loads for crops and forests. Dose-response functions are used to quantify the impact on human health and the ecosystems. The results show that the costs of premature deaths and morbidity cases caused by urban air pollution are an order of magnitude larger than the losses of crops and forests.

Benefit-cost comparisons of control options suggest that low sulfur coal holds the largest potential in reducing the health and ecosystem damages associated with urban emissions. Using cleaner coal is especially attractive because cities in China are only likely to reduce their dependence on coal gradually. Regulatory lessons from Shijiazhuang and Changsha suggest that market-based policy initiatives, such as coal price reform and emission levy, to adequately internalize the externality of the health and environmental costs associated with dirty coal is critical to encourage the switch to clean coal.

Market-based incentives are also important to support further progress in energy conservation to reduce fuel consumption and emission. By removing market distortions and providing information to energy users, saving energy appears to be the most efficient among policy option. But conservation alone will not prevent further large increase in future emissions. The penetration alternative fuels such as natural gas and the control of small emission sources must be integrated in a comprehensive abatement strategy. In order to reduce ambient concentrations to meet air quality standard and to reduce acid deposition to be below the critical loads to minimize the risks for ecosystems, China will have to employ such an integrated strategy in various cities as contemplated in the two control zone policy.

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Figure 15.1 China's Two Control Zones and the Selected Case Study Cities

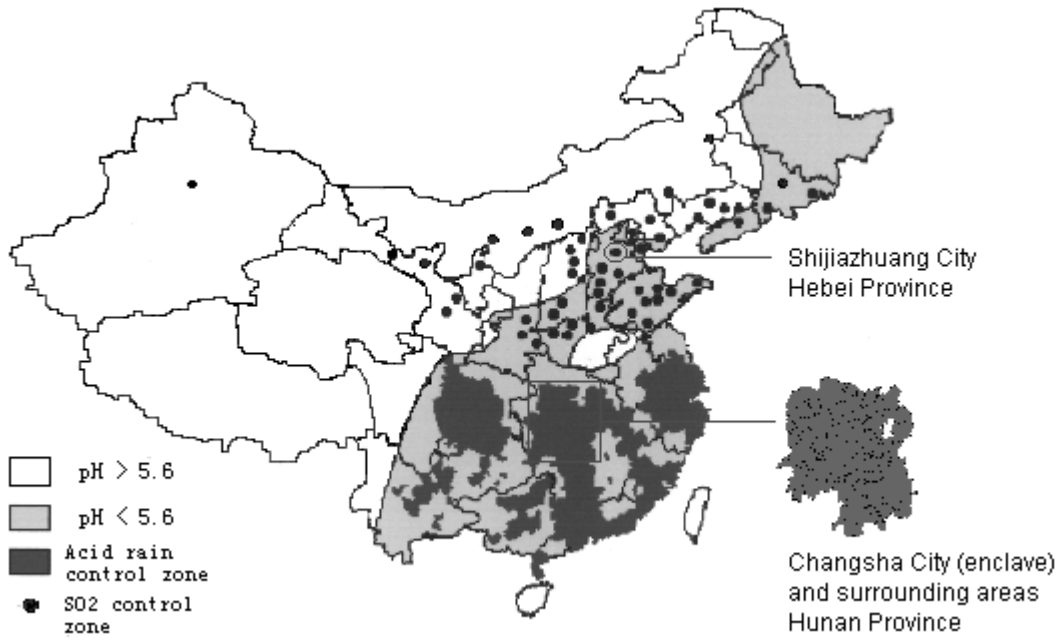


Figure 15.2 A Framework for Modeling Urban Air Pollution

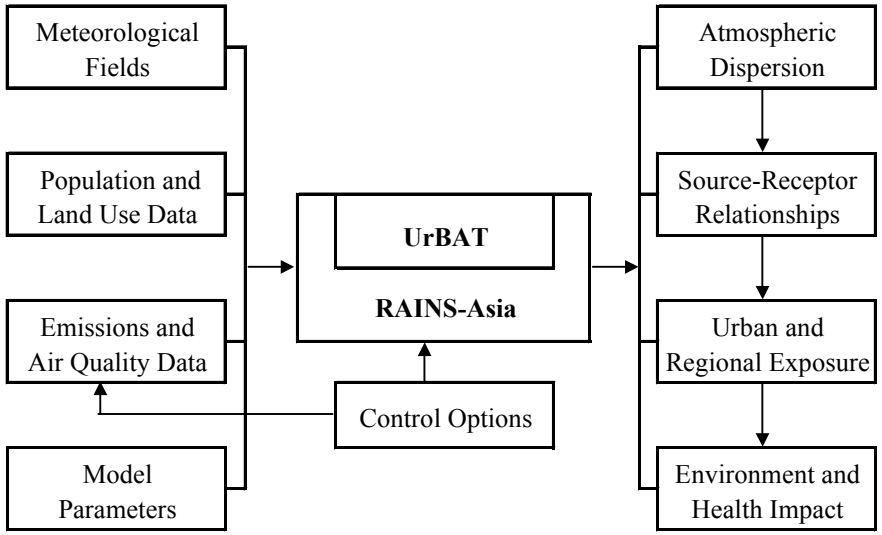
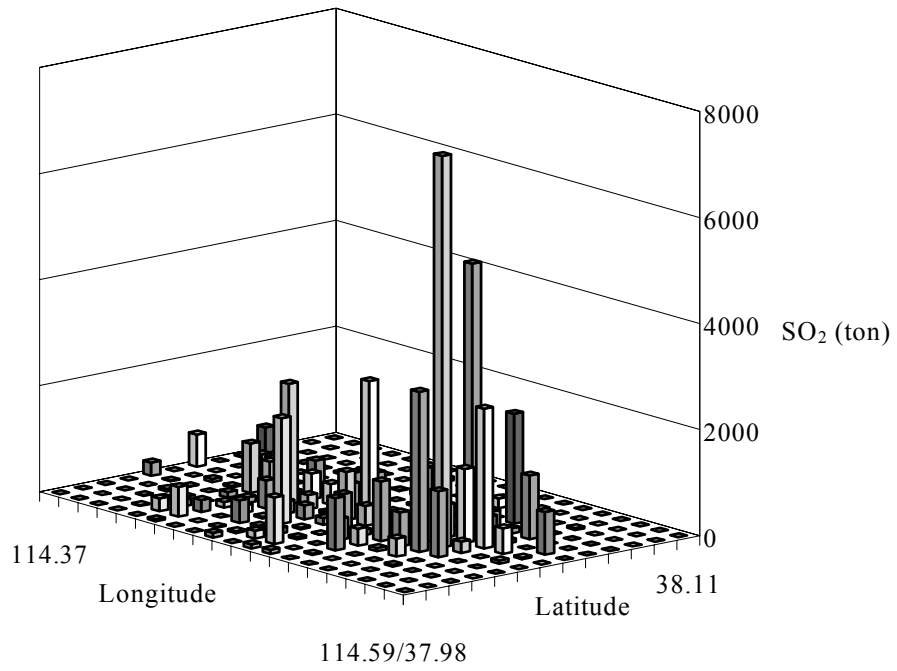


Figure 15.3 SO₂ Emissions in Shijiazhuang in 1 km by 1 km Grid Cells (Tons), 2000



Source: Data collected from the World Bank missions to Shijiazhuang, August 2000 and July 2001.

Figure 15.4 Annual Average SO₂ Concentrations in Urban Shijiazhuang, 2000

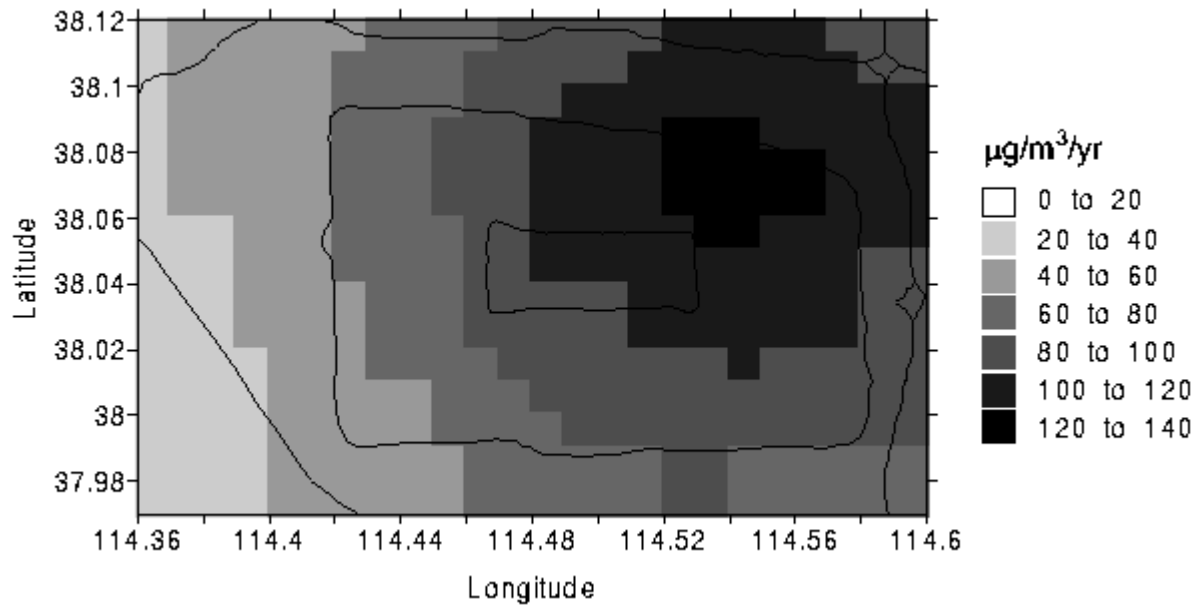


Figure 15.5 Annual Average Sulfate Concentrations in Urban Shijiazhuang, 2000

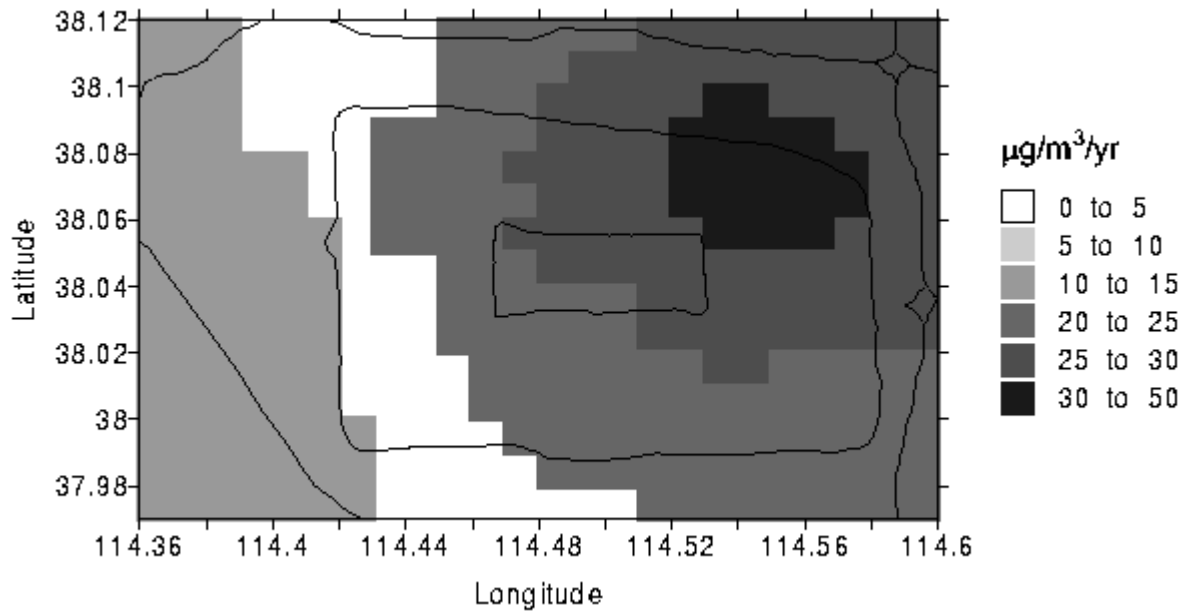
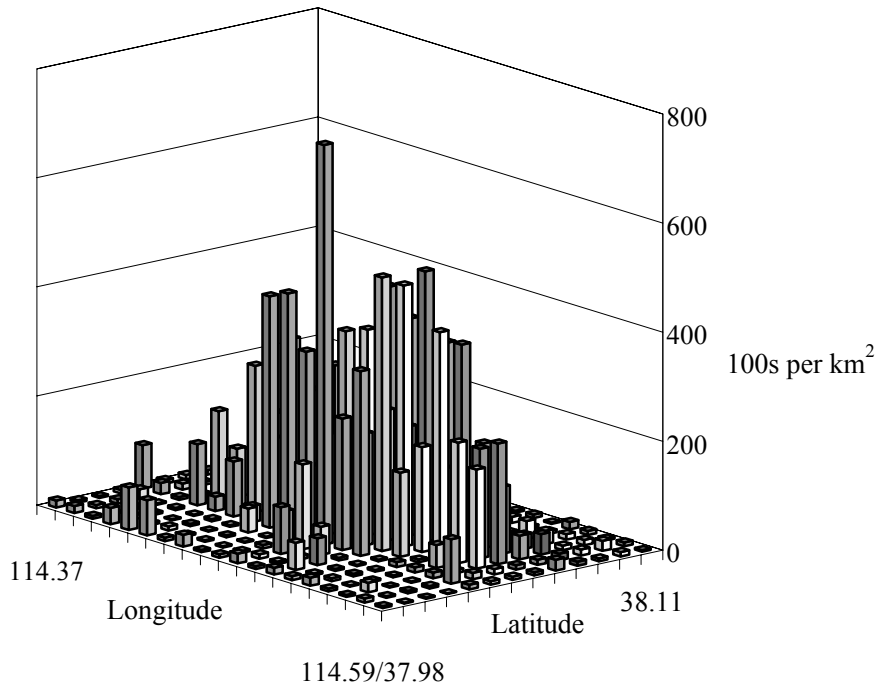
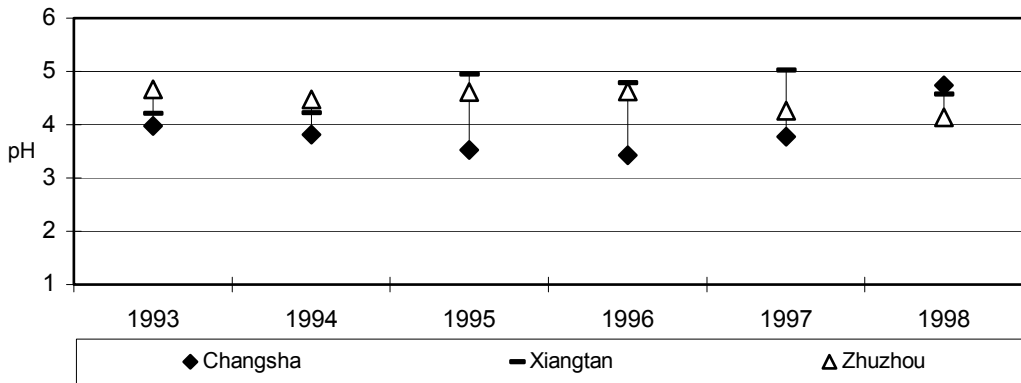


Figure 15.6 Population Density in Shijiazhuang in 1 km by 1 km Grid Cells, 2000



Source: Oak Ridge National Laboratory (2001), LandScan 2000: Global Ambient Population, 2000.

Figure 15.7 Acid Rain pH Value in Changsha, Xiangtan and Zhuzhou



Source: Data collected from the World Bank missions to Changsha, Xiangtan and Zhuzhou, August 2000 and July 2001.

Figure 15.8 Acid Deposition Originated From the Changsha Region, Year 2000

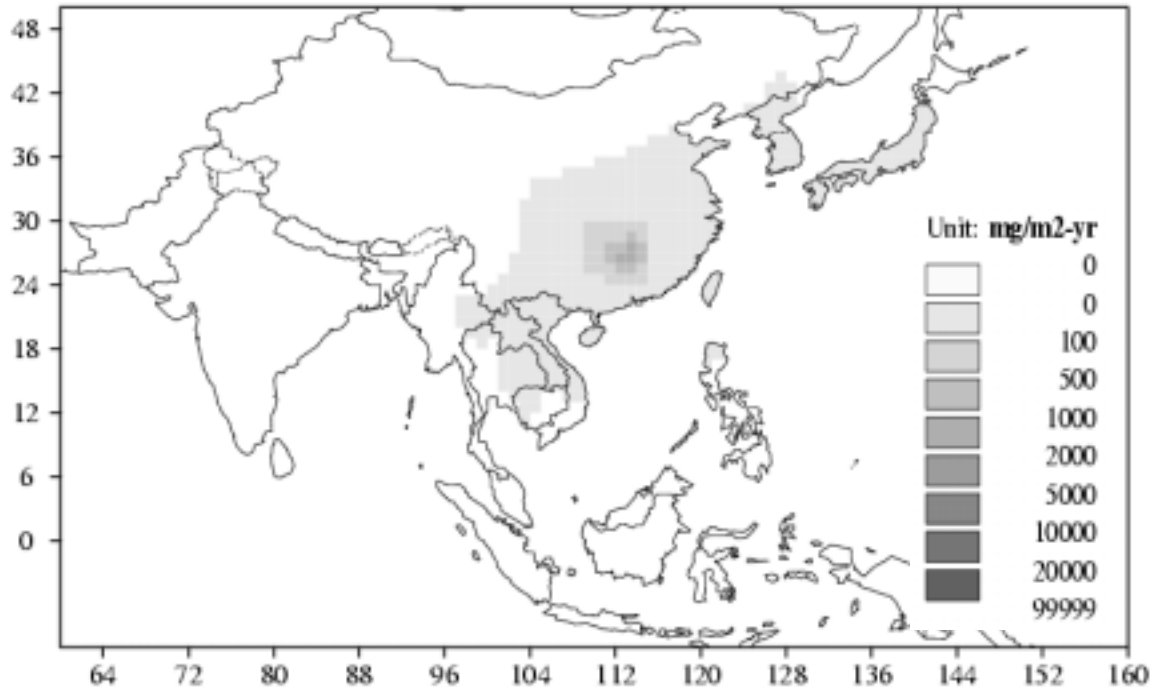


Figure 15.9 Excess Acid Deposition Above Ecosystems Critical Load, Year 2000

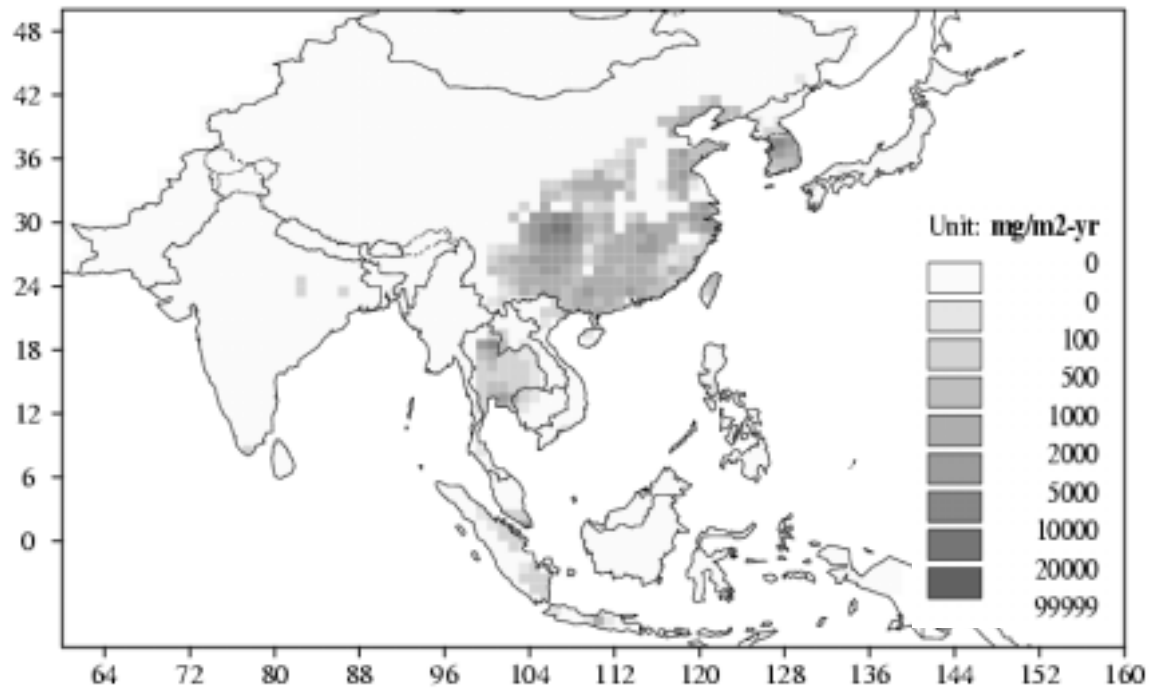


Table 15.1 Chemical Analysis of Rain Samples, Changsha, 1991-1997

	1991	1992	1993	1994	1995	1996	1997
<i>pH</i>	3.79	3.87	3.84	3.73	3.49	3.42	3.74
H^+	162.18	134.9	134.9	186.21	323.59	380.19	181.97
Ca^{2+}	152.69	246.01	102.3	106.79	164.79	105.79	178.14
NH_4^+	88.64	77.01	77.01	87.53	88.96	63.38	58.94
NO_4^-	33.55	21.61	26.12	27.41	25.76	26.89	20.93
SO_4^{2-}	325.62	255.25	187.8	202.17	270.4	144.77	186.58
Mg^{2+}	11.52	23.87	11.52	18.93	18.93	9.89	9.05
K^+	12.02	24.04	5.63	6.91	7.68	3.58	4.1
Na^+	32.62	59.16	3.91	6.09	7.4	2.18	2.61
F^-	-	20.73	13.16	8.42	6.31	6.84	6.84
Cl^-	12.13	16.08	14.67	11	12.41	37.22	12.41
$\frac{SO_4^{2-} + NO_3^-}{Ca^{2+} + NH_4^+}$	1.49	0.86	1.19	1.18	1.17	1.01	0.87
$\frac{SO_4^{2-}}{NO_3^-}$	9.71	11.81	7.19	7.83	10.5	5.38	8.91

*Chemical indexes are measured in ueq/l, except for pH and the ratios in the last two rows.

Source: Data collected from the World Bank missions to Changsha, August 2000 and July 2001.

Table 15.2 Health Effects of Sulfate Pollution, Shijiazhuang, 2000

	Dose-response Coefficients*	Health Impacts	Economic Valuation	
			US\$ per unit	Million US \$ in total
Mortality				
Premature deaths	6	251	160000	40.15
Morbidity (cases)				
Acute				
Respiratory hospital admissions	12	502	284	0.14
Emergency room visits	235	9829	23	0.23
Lower respiratory infection/child asthma	23	962	13	0.01
Asthma attacks	2608	109076	4	0.44
Respiratory symptoms (thousand cases)	183	7654	0.6	4.59
Chronic				
Chronic bronchitis	61	2551	8000	20.41
Restricted activity days (person year)	157.5	6589	846.8	5.58
Health cost in total (million US\$)				71.5
Health cost as % of local GDP [#]				4.3%

*Effects per 1 million people for every 1 $\mu\text{g}/\text{m}^3$ increase in ambient concentrations of PM_{10} (World Bank, 1997, p. 24)

[#] Derived from per capita GDP. Total GDP for Shijiazhuang municipality in year 2000 was RMB 100.1 billion, and urban residents were about 14% of the municipal population. The exchange rate was 8.34 RMB/US\$ for year 2000.

Table 15.3 Benefit-Cost Comparisons between Pollution Control Options, China, 2020

	Benefits (US\$ billion)		Benefit/Cost Ratio	
	Health Effect	Crop/Forest Impact	Health Effect	Crop/Forest Impact
1. Low sulfur coal	54.3	3.4	6.6	0.41
2. Small sources abatement	43.7	2.6	8.1	0.48
3. Energy conservation	31.8	1.8	197.0	11.08
4. Fuel substitution	13.3	0.7	1.7	0.09

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