
Air Pollution and Health Damages in China: An Introduction and Review

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1.1 Introduction

By some assessments, one of the leading causes of death in the People's Republic of China is respiratory diseases caused by air pollution. Although some would debate this claim, few would question that air pollution is a very serious problem in China. High-level concentrations of particulates and sulfur dioxide (SO₂) are recorded in many cities, levels that rank a number of them among the worst polluted in the world. This pollution is produced by a variety of sources, including a burgeoning transportation sector, residential heating, and light industry. Much of it is generated by power plants and heavy industry, often located in urban areas by industrial policies predating the post-1978 reform era.

The authorities have responded to this challenge, and there has been clear progress in some areas of pollution control. As a result of these actions, and concurrent changes in economic policies and the structure of the economy, concentrations of SO₂ and total suspended particulates (TSP) declined in many places, especially during the 1990s. For instance, total SO₂ emissions fell 10% between 1995 and 1999, and the averaged ambient concentrations in thirty-two major cities declined from 100 µg/m³ (micrograms per cubic meter) in 1991 to 62 µg/m³ in 1998 (World Bank 2001). Another compilation of official data reports that annual average TSP across 140 Chinese cities fell from a mean of 500 µg/m³ in 1986 to 300 µg/m³ in 1997 (Florig et al. 2002).

Despite these improvements, the downward trends appear to have largely abated or reversed, and concentrations remain higher than national standards and international guidelines in many localities. The Chinese standard for annual average TSP in residential areas is 200 µg/m³, for instance, and the World Health Organization's (WHO) guideline was 90 µg/m³. We should note that the WHO has now stopped setting guidelines for TSP, though it has begun to set guidelines for finer particles,

the fractions of TSP that are considered most harmful to health (WHO 2002, p. 186; WHO 2005). Pollutant concentrations are increasing in some major cities. Estimates of the value of total air and water pollution damages nationwide have ranged from 3% to 7.7% of gross domestic product, usually dominated by the air pollution component. Although these estimates are rough and involve subjective judgments, few would dispute that the damages are large compared to China's own past and compared to more developed countries. Furthermore, these damages may well rise with increasing energy use from China's vigorous economic expansion.

We need to think beyond the ambient pollution levels to consider what ultimately concerns us about China's degraded air quality: the damage it causes. These include impacts on human health, on materials such as built structures, on natural resources such as forests, on the productivity of crops, and on ecosystems. All of these, in turn, affect the economy. The assessment of this volume focuses on refining understanding of the damages to human health, which most researchers believe dominate the total impacts. The other, lesser damages are topics for future extensions of this work.

With a focus on health risk, we must keep in mind that what concerns us is not the level of pollution itself, but rather the level that reaches human lungs—a distinction that fundamentally motivates and shapes this assessment. We must consider such factors as the rapid urbanization in China and how expanding cities and rural-to-urban migration locate an increasing proportion of the Chinese citizenry nearer to pollution sources. In terms of total population risk, this could even outweigh whatever gains in pollution control have occurred. The crucial factor of human *exposure* is not often considered carefully in pollution damage assessments in China.

Whereas it is generally accepted that Chinese health and economic performance have been harmed by its degraded air quality, the power of many existing damage estimates to influence policy has been undermined by their uncertain and aggregate nature. More reliable and comprehensive estimates based on detailed energy use and emissions could help guide environmental policy and law and orient public and official opinion to expanded preventive action. Disaggregating the estimates to specific industries, or geographical areas, can help target policies.

To make such detailed estimates, the prevailing approach in more developed countries has been (1) to use air-dispersion models to characterize the link between emissions of all sources and atmospheric concentrations of pollutants in a target region; (2) to estimate human health impacts on the basis of exposure-response functions; and (3) to monetize these damages by using some valuation method. A large

body of such research has been developed over the course of many years and at considerable expense in many nations.

Researchers in this field in China, however, face major limitations compared to those in the West. A key challenge is that basic pollution data are sparse, because the national emission inventories and monitoring infrastructure are still developing. Local institutions have only recently begun to use advanced air-dispersion models, and have applied them comprehensively—that is, to estimate all sources—for a few localities only. For example, the State Environmental Protection Administration (SEPA) and the U.S. Environmental Protection Agency (U.S. EPA) have in recent years commissioned studies using such models and exposure assessments in major cities such as Shanghai (Shanghai Academy of Environmental Sciences [SAES] et al. 2002) and Beijing (Tsinghua University et al. 2005). For health valuation, such studies have had to try to translate Western results to Chinese conditions or use results from one of just a few small unpublished studies completed in China. As we will describe in section 1.6, a small number of additional international collaborations with Chinese institutes have used these approaches to investigate the impact of air pollution in other cities and in provinces. Additional studies have investigated specific economic sectors, such as electric power generation.

All of these assessments provide valuable information and are useful to informing and improving control strategies. Few of them, however, identify *national* priorities, evaluating the best use of the pollution control yuan across the entire economy. To set national pollution-control priorities, the Chinese government has had to rely on simplified analyses and rules of thumb rather than comprehensive assessments that permit systematic benefit-cost analysis.

As we will summarize later, a World Bank report from 1997, *Clear Water, Blue Skies*, was a first international collaboration to estimate aggregate damages from air and water pollution, as high as 7.7% of Chinese gross domestic product (GDP). This provided decision makers with a useful estimated scale of damages, but its methods were debated in China and beyond and, in any case, were understood to provide rough approximations. Furthermore, it was limited for national priority setting because it was not designed to identify sectors that contribute most to this pollution or to specify impacts of alternative control policies systematically on both the public health and the economy.

The effort of this volume was conceived to build, and improve, on this important first effort by the World Bank with a more extensive and deeper air pollution damage assessment. We believe it also complements several ongoing collaborations involving SEPA and its affiliated bureaus and research entities, Tsinghua and other

universities, and a number of international partners. The latter include the Center for International Climate and Environmental Research—Oslo (CICERO), the Organization for Economic Cooperation and Development (OECD), the World Bank, and the U.S. EPA. It contributes to the body of policy-targeted air pollution research in a number of ways that are also intended to advance and promote development of the scholarly literature in this area:

- by developing more emission data and air-dispersion estimates on the basis of local meteorological conditions and actual source characteristics;
- by applying a new method to make a best-available assessment, given data limits, of national human exposure to key ambient air pollutants from major sectors;
- by conducting one of China's first systematic contingent valuation studies to help build a Chinese health valuation literature;
- by providing a framework for integrated analyses of economy-wide costs and benefits of air pollution control policies; and
- by conducting such analyses of two control policies: damage-weighted taxes on fuels and sector outputs.

We do not seek the greatest possible precision in the damage assessment, which would take far more effort and resources than we could provide. We aim, however, to make reasonable estimates of the main sources of pollution, to permit inter-industry comparisons (and not just estimate aggregate damages, as in the above-mentioned World Bank study), and to create a basis for more detailed evaluation as data and research capacities improve. Our health damage estimates are directly linked to economic activity and energy use on a sector basis, allowing us to identify the sources of damage and allocate responsibility. This can help to prioritize pollution sources for emphasis in national energy and emission control policies, and to understand their effects throughout the economy.

Our assessment focuses on particulate matter (PM) and SO₂ in determining health effects in China, for two reasons. First, we believe that they likely dominate other pollutants as the source of air pollution health damages in China. Second, all such studies are limited by research practicalities, and the data for estimating the effects of PM and SO₂ are more readily available in China—though not necessarily simple to obtain—than for other pollutants. It is nevertheless important to note other pollution types as additional concerns for control and future research. In particular, as the transportation sector continues its explosive growth, we expect the health risks associated with nitrogen oxides and ozone to grow swiftly. China has already made impressive progress against a third mobile-source pollutant, lead, with a national ban on production and sale of leaded gasoline that became effective in 2000.

We further note the wide awareness that emissions of local air pollution and global greenhouse gases (GHGs) are closely related and ideally should be studied and addressed jointly. Our module for benefit-cost analysis explicitly evaluates both local pollutants such as PM and SO₂, and the primary anthropogenic GHG, carbon dioxide. This allows us to assess a benefit to the global environment of reducing China's local air pollution.

1.2 How to Read This Book

This book is explicitly targeted at a wide range of audiences, from curious lay readers, to informed nonspecialists and policy makers, and ultimately to researchers and scholars with either interdisciplinary or more narrowly specialized interests. The structure of the volume reflects this effort to appeal to many communities. The authors themselves are environmental health scientists, engineers, and economists. The research detail and methodological presentation deepen as the book progresses, from the introductory (chapters 1 and 2) to the specialized (chapters 5–10), with two transitional chapters in between (chapters 3 and 4). The scope of the book also evolves, because it addresses a consummately interdisciplinary topic by building up from a set of more focused studies. It begins with a broad, summarizing perspective (chapters 1–3), proceeds through more narrow ones (chapters 4–8), and then returns to the wider, integrated scope at the end (chapters 9–10).

Readers should approach the book accordingly. Nonspecialists should focus first on part I, including the introduction and background presented here in chapter 1 and a summary of the research project and its conclusions for policy purposes in chapter 2. If their interest is piqued, they can gain a much deeper understanding of the assessment by at least beginning part II, which presents the research itself. Chapter 3 is specifically written for nonexpert readers as well as expert ones, and as a transition between parts I and II. It repeats the scope of chapter 2, summarizing the project and its conclusions, but much expanded to its full methodological and research context. (In fact, chapter 2, as a summary for policy, is mainly just an abbreviation of chapter 3.) We hope that many lay readers will want to read this far, to learn about the nature of these types of studies and how they produce their conclusions. (Environmental assessments, after all, are often used in public and political debates about environment and can become the basis for lasting government policies. They are sometimes reported in popular media—imagine a headline: “Study Says Air Pollution Kills 1,500 Per Year; Costs Estimated in Billions”—and

it is useful to know how to think critically about them.) At the conclusion of chapter 3, readers will have a sense of the remaining chapters that might most interest them.

Specialist readers and scholars may be more interested in the academic studies at the core of the assessment and may quickly choose to focus their attention on part II. They can treat chapter 3 as their introduction to the integrated aims of the project, chapter 4 as an introduction to central human exposure and epidemiological issues in the book, and then proceed through the original research of chapters 5–10 as their interests dictate. These chapters are written in detail so that our methods and results, including their limitations, are clear and can inform future research. They are still meant, however, for engaged readers of diverse expertise, with limited disciplinary jargon and introductory sections. The most specific technical and methodological details are placed in appendices. This is not to suggest that part I and this chapter offer specialists nothing, especially if they are unfamiliar with China's energy and air-quality conditions or with the existing literature on the topic. Researchers may want to skip chapter 2 altogether, however, because they will find chapter 3 a much more informative version of the same material.

The remainder of this chapter provides background on the topic, introduces our analytical approach, and summarizes related analyses by others to date. Specifically, sections 1.3 and 1.4 describe recent energy use and air pollution conditions for those who may be unfamiliar with China. In section 1.5, we outline how our research investigates health damages of air pollution and analyzes policies to reduce them. In section 1.6 we briefly mention other studies that have been conducted in this field, from which we have gained important insights for our own program.

1.3 Energy Use in China

Two prominent features of China's energy structure warrant a brief overview. One is China's very high, and continuing, dependence on coal. The second is a swift decline in energy intensity since economic reforms began in 1978.¹

Economic growth has far exceeded growth of energy consumption in the reform period since the late-1970s, as shown in figure 1.1. In the ten years prior to 1997, the primary energy consumption per yuan of GDP fell by 37%. If one accepts the somewhat questionable official figures for the late 1990s and early 2000s, showing a sudden decline in coal use in that period (discussed below), it fell another 26% by 2000, before leveling and then rising (Sinton et al. 2004, table 4B.2). (Many statistics reported in this section are from this source, which compiles energy-related data from official sources and reports them in English. Table and figure numbers are

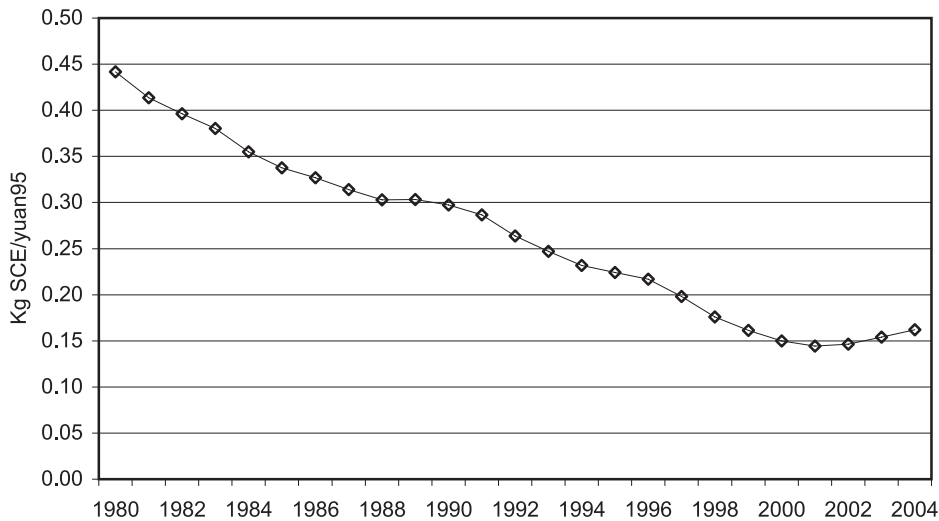


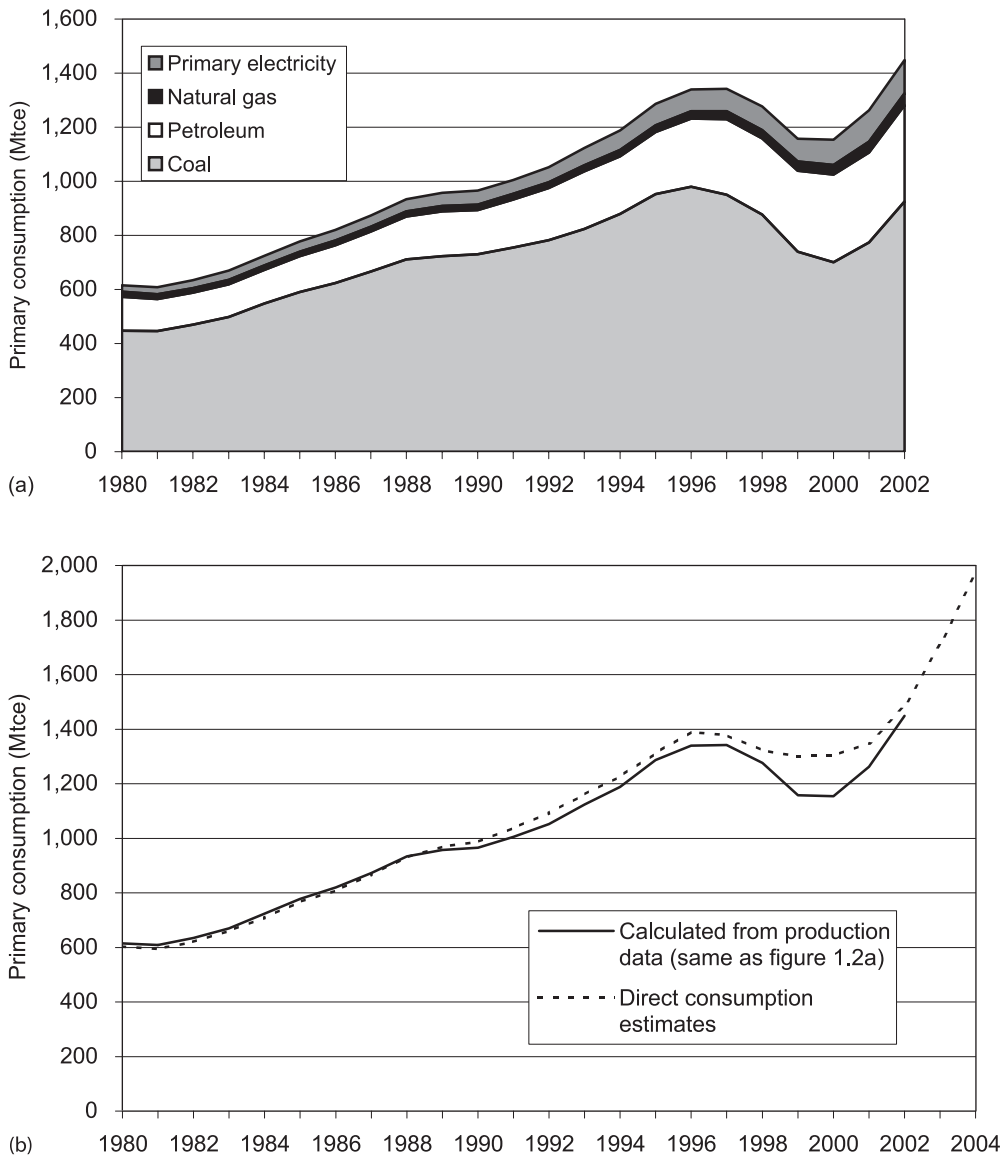
Figure 1.1

Energy consumption per unit GDP. Total primary energy is the sum of coal, oil, gas, and hydroelectric power, all converted to kilograms of standard coal equivalent (SCE) units. Yuan95 is yuan in constant 1995 value. Source: Sinton et al. 2004, table 4B.2.

included so it can be consulted for citations of the original data sources of the Chinese government.)

Despite this remarkable overall transformation, China remains one of the most highly energy-intensive major economies in the world, defined by the amount of energy needed per dollar of output. That noted, China's energy use on a per capita basis remains exceedingly low compared to more developed countries. In 2000 energy use per capita in China was about one-twelfth that of the United States, and about one-sixth that of Japan (U.S. EIA 2003).

Total energy consumption by source is represented over time in figure 1.2a. China is the world's largest producer of coal, generally the worst fossil fuel for both local and global pollutant emissions. Coal supplied 70–76% of its commercial energy consumption from 1980 until 1997, after which the disputed data suggest a share decline to 61–64%. The main users of coal are power generation and industry (mostly manufacturing), accounting for about 48% and 35%, respectively, in 2002² (Sinton et al. 2004, tables 4A.1.4 and 4A.17.2). According to official data, annual consumption of coal exceeded 1300 million tons during 1995–1997, fell sharply to 982 million tons in 2000, and rose again to 1579 million tons by 2003 (NBS 2005).

**Figure 1.2**

Primary energy consumption in China. Consumed energy is converted to million tons of standard coal equivalent (Mtce) units. Primary electricity is power generated from sources other than combustion of fossil fuels. It is chiefly hydroelectricity and a small amount of nuclear power. (a) Consumption calculated from production data. (Source: Sinton et al. 2004, table 4A.1.2.) (b) Comparison to direct consumption estimates. (Source: NBS 2006, table 7-2.)

Concerning the data in the late-1990s into 2001, the dramatic dip in coal consumption shown in figure 1.2a came at the same time that official statistics show the economy growing at more than 7% per year. If this is accurate, it is an historic event in decoupling economic growth from energy demands.

Sinton and Fridley (2000) reviewed a number of reasons that might explain such a decline, including closure of small factories and changes in economic structure, gains in end-use efficiency, fuel-switching, a rise in coal quality, and a policy of the central government to close small mines. In light of a high-profile, international spotlight on the inferred environmental benefits of this sudden downward dip in coal use, including for greenhouse gases—e.g., articles in *Science* (Streets et al. 2001) and the *New York Times* (Eckholm 2001)—we note that a wide debate about the accuracy of China's official data for the late-1990s promptly followed. A variety of doubts about the energy and economic statistics have been described (Rawski 2001; Sinton 2001; U.S. Embassy 2001). Others have continued to propose economic and other explanations for this remarkable trajectory of energy consumption, such as transformations occurring in anticipation of China's accession to the World Trade Organization (Fridley et al. 2003; Fisher-Vanden et al. 2004).

We will not belabor this debate here but will note that the National Bureau of Statistics (NBS) includes a balance term in its official coal tables that represents the difference between direct estimates of consumption and those based on production data. In principle, these will be equal; the latter are more often reported because they are generally considered more accurate. The size of this residual term grows somewhat in 1996–1998 compared to earlier years in the decade and then rises dramatically for the years 1999–2001 (three-fold and more, to *hundreds of millions* of tons), falling back to the previous scale in 2002.³ Thus the direct consumption estimates (NBS 2006) show a shallower dip in 1999–2001 compared to the production-based data of figure 1.2a, as shown in figure 1.2b. Nevertheless, even this alternative measure shows an unusual decline in coal consumption at a time of strong overall growth in the economy.

The reporting of the ballooning balance term indicates that the NBS is not unaware of growing inconsistencies in its coal data for those years but, rather, acknowledges and indeed quantifies them. The cause of these data problems is certainly of pertinent interest to the government; a contributor may be disruptions from reform of systems to compile and report statistics under the broader government reorganization in 1998 and as the objectives of data provision continue a shift from support for a planned economy to informing a market one. It would be unsurprising if this process encountered difficulties in implementation in view of the size

and inertia of this system, and we assume that efforts to improve the statistical infrastructure continue today. Regardless of possible explanations, for purposes of research we take the NBS at its word about the existence of large statistical inconsistencies for this period. For this reason we are cautious about using national data from the years in which this residual term is anomalously large—1999–2001—to indicate trends in coal use and related emissions over time.

Another important trend in China's energy structure is the growth in petroleum consumption and imports, driven by a rapidly motorizing transport sector. From 1990 to 2002, the national stock of passenger vehicles increased more than 18% annually, to 12 million units, and motorcycles increased an average of 21% annually, to 43 million units. This contributed to a growth in petroleum consumption of 6.7% annually over these years and an increase in petroleum share of total primary energy from 16.9% to 24.6%. China is a significant petroleum producer but became a net importer in 1996, and by 2003 net imports of crude oil had reached 83 million tons (Sinton et al. 2004, tables 5B.3, 4A.1.4, and 7A.1.1).

China is usually described as geologically limited in natural gas resources, though this energy resource was also historically underexplored and underdeveloped. Driven in part by air-quality concerns, the government in recent years has promoted the extraction and distribution of this cleanest of fossil fuels, particularly for domestic heating in major cities. A major pipeline from gas fields in western China to central and eastern demand centers is partially completed. Even with the recent government efforts, however, gas comprised only 3% of total primary energy consumption in 2002 (Sinton et al. 2004, table 4A.1.4).

Other sources of commercial energy in China include substantial hydropower, which provided 17.6% of gross electricity generation in 2002, and nuclear, which provided 2.3%. These two constituted 9.1% of total primary energy production in 2002. Biomass—chiefly crop wastes and firewood—is a major fuel in rural China and a source of indoor air pollution, but it is not generally a commercial fuel and is left out of most energy tables. When counting all sources of energy, not just commercial forms, biomass constitutes 15% of total energy, compared to 56% by coal and 20% by petroleum in 2002 (Sinton et al. 2004, tables 2A.4.2, 2A.1.3, and 4B.1).

Although they are somewhat imposing, we offer two more figures on energy, because they allow the reader to appreciate the relative scales of both the sources of energy and its end uses. Figure 1.3 represents the entire energy system of China, with flows translated into equivalent units across energy types and scaled accordingly. We also include a similar diagram for the United States (figure 1.4) to contrast

China's energy structure with that of the largest developed country. The end-use categories are not precisely the same, reflecting different conventions in economic and energy data in the two countries. We also caution that construction of such diagrams requires many assumptions, especially in estimates of the efficiency of conversion into "lost" versus "useful" energy. One should view these diagrams as representations of the big picture of energy flows in the two countries rather than sources of detailed information.

The tremendous coal dependence in China compared to the United States, and the minimal role of natural gas, is clear in these figures. Petroleum has grown to a sizable share of total energy use in China, though still lower than richer countries such as the United States. Notable for China on the consumption side is the huge share of energy consumed by industry, via direct combustion of petroleum and especially coal, along with electric power. China's transportation sector consumes a relatively small share of total energy, despite its enormous recent growth, whereas it is the largest end-use in the United States. Hydroelectricity provides a larger share of energy consumed in China than in the United States, whereas the reverse is true for nuclear power. We must spotlight the role of biomass in the form of crop wastes and fuel wood as a sizable energy source in rural China, repeating the caveat that this flow is noncommercial and left out of most energy tables for China, so one should not assume that these data are comparably collected.

Finally, we should note that, if the scaling were consistent across the two diagrams, the U.S. flows would collectively be roughly one and a half times the size of China's, to fuel an economy that was eight times the size (unadjusted for purchasing power) and to support a population less than one-fourth as large.

1.4 Air Pollution Emissions and Ambient Concentrations

As mentioned above, China's air quality is generally poor, with two-thirds of 338 monitored cities out of compliance with at least one of the nation's air-quality standards for residential areas in 1999 (World Bank 2001). We limit this section to a brief summary of key characteristics and trends in the emissions and ambient conditions for the pollutants that are the subject of the current study.⁴ A more detailed discussion of China's air pollution is PRCEE et al. 2001, a background study for World Bank 2001.

We note an important distinction about these data: the emission data are for the entire country, whereas the ambient concentration data are only for major cities. In any case, the link between emissions and concentrations are more complex than

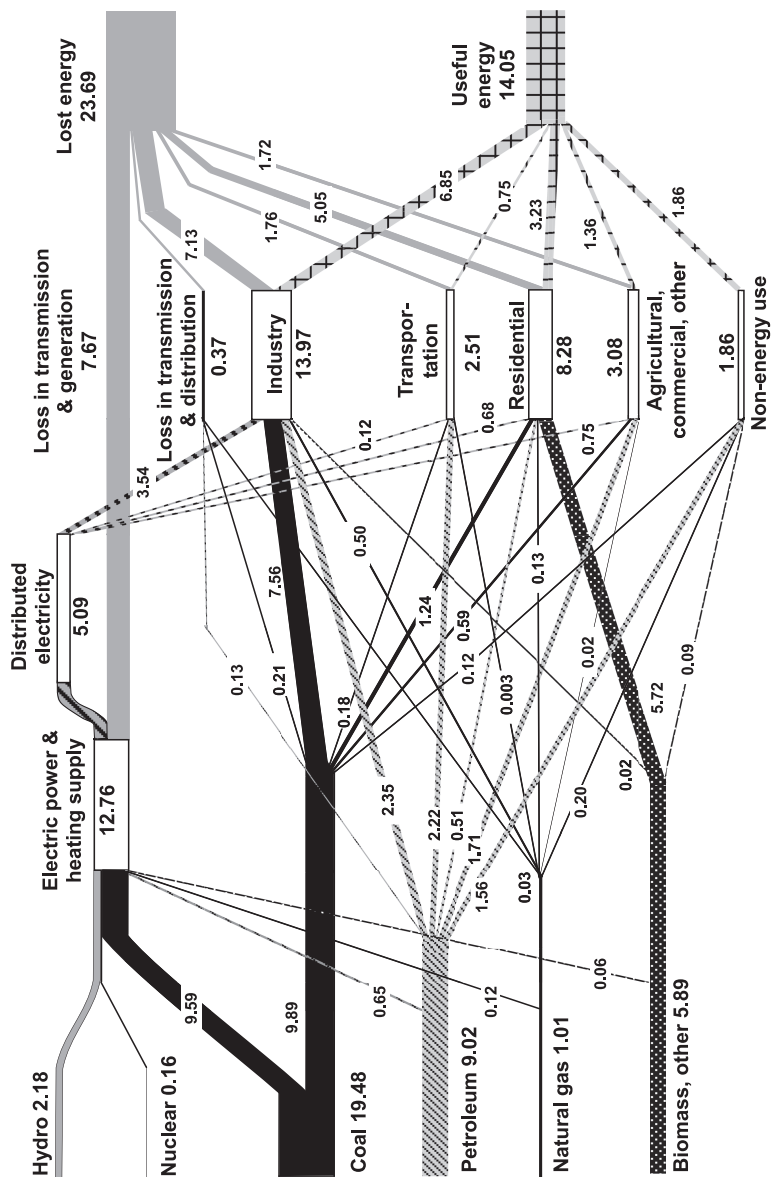


Figure 1.3

People's Republic of China energy flows for 2000: Total primary energy consumption of approximately 37.75 quadrillion British thermal units. Source: NBS 2004. These data were compiled and the figure was first drafted for the authors by Qiaomei Liang, Yiming Wei, and Ying Fan of the Institute of Policy and Management, Chinese Academy of Sciences.

many realize. Investigating this is one of the primary motivations of the research of this volume, as there are many factors that can influence atmospheric dispersion, removal, and transformation of pollutants after they are emitted.

1.4.1 Particulates

The SEPA data on particulate matter have, until recently, only systematically covered TSP. There has been less information on finer particulates such as PM_{10} or $PM_{2.5}$ (particles less than 10 or 2.5 microns in diameter, respectively), though reporting of PM_{10} ambient concentrations has increased in recent years. These are the forms most closely associated with adverse health effects and that most current particulate epidemiology investigates. For health studies in China the long reliance on the TSP measure has necessitated an estimation of the fraction that is PM_{10} and/or $PM_{2.5}$.⁵ TSP is classified as combustion emissions (“soot” in official publications) or process emissions (“dust”).⁶ The coverage of this TSP data is being improved, with an effect that historical data are not comparable over long periods. Earlier data included just medium and large enterprise sources, and only in the last decade were the environmentally significant township and village enterprises (TVEs) gradually added.

National particle emissions from combustion sources are given in figure 1.5 (SEPA 1992–2004), where the break in 1995 reflects the inclusion of data from the

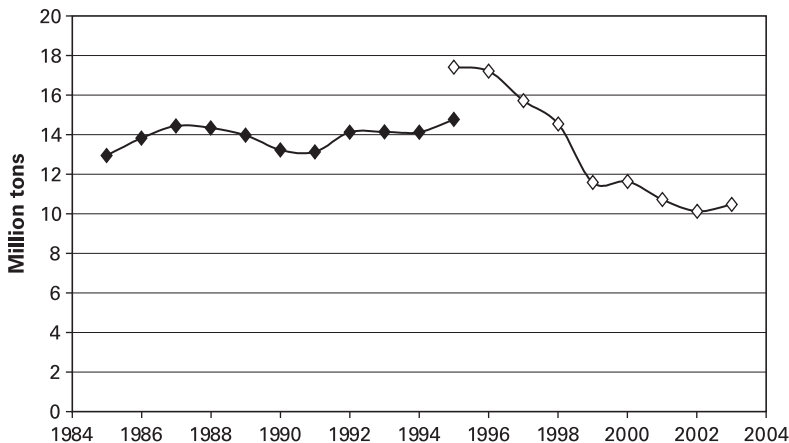


Figure 1.5

Combustion emissions of total suspended particulates in China. Data coverage extended in 1995 to include township and village enterprises. Source: SEPA 1992–2004.

TVEs. The data show stability in the early 1990s and a considerable reduction since 1995. When noncombustion emissions are included, the initial level is doubled, but the downward trend is sharper because of the more rapid reduction in process emissions. We are not aware of a systematic study of these trends, but it appears clear that improved pollution control played a role in the decline in emissions over the 1990s, on top of the effects of the slowing of total energy consumption shown in figure 1.2.

Particulate emissions by sector are reported later in chapter 9, table 9.1. The combustion emissions are mostly from electricity generation (32% of the total in 1997) and cement and related products (21%). The primary fuel source is coal. Process emissions are largely from cement production and iron smelting, at roughly 70% and 15% of the total, respectively.

Urban ambient concentrations of TSP result from the above emissions and secondary transformation of other pollutants, such as sulfur and nitrogen oxides (NO_x) described below. SEPA compiles data for cities, contrasting northern and southern ones, and they are reproduced in figure 1.6. On the basis of these cities, the national average TSP concentrations for urban areas declined about 30% from 1990 to 1999.

Although this trend was encouraging, it has leveled since 1999, with the average concentration in 2003 reaching $256 \mu\text{g}/\text{m}^3$. As important, the national TSP standard for residential areas (termed “class II”) is $200 \mu\text{g}/\text{m}^3$, and the WHO guideline was $90 \mu\text{g}/\text{m}^3$ until it was suspended for this particulate form. About 60% of the surveyed

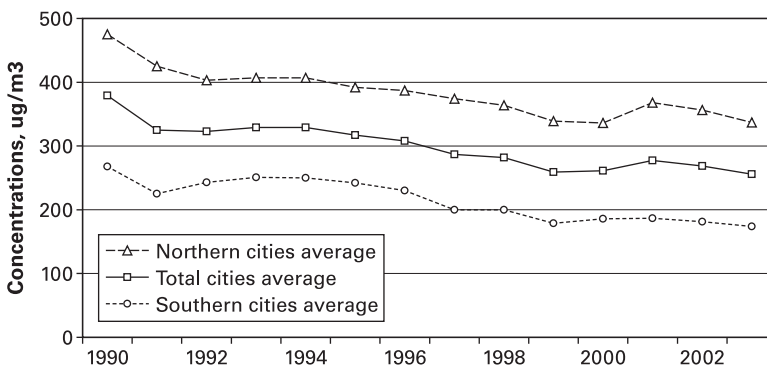


Figure 1.6

Average ambient concentrations of total suspended particulates in cities of China. Source: Sinton et al. 2004, SEPA 2004.

Table 1.1Annual average measured PM₁₀ concentrations in major cities for 2003

	Number of Cities	Annual Average Concentrations (µg/m ³)
Northern cities	52	140
Southern cities	54	102
Total cities	106	121

Source: SEPA 2004.

cities in 1999 exceeded the Chinese class II standard. Indeed, 36% exceeded the class III standard, designed for industrial areas (PRCEE et al. 2001). The figure also illustrates a marked difference in ambient TSP conditions between southern and northern cities, the southern ones, on average, meeting the class II standard in recent years with northern ones still far from compliance. It is difficult to explain the higher TSP levels in the north. It may be obvious that the colder north requires heating, but one also has to consider the effect of higher background concentrations and wind-blown dust in addition to human emissions.

As noted above, there was for a long time little information available on the smaller particulate matter that is known to be most harmful to health. Since 1999, ambient PM₁₀ has been subject to an official class II (residential) air-quality standard of 100 µg/m³, and hourly monitoring of it is now carried out in many cities, with annual averages now reported in official sources in addition to TSP. The recently announced WHO annual average guideline is 20 µg/m³ (WHO 2005). In table 1.1 we summarize annual average PM₁₀ concentrations for major cities in 2003, again classified as either northern or southern (SEPA 2004). Seventy-eight percent of northern cities and 52% of southern ones showed annual averages above the class II standard.

1.4.2 Sulfur Dioxide

The officially reported national SO₂ emissions are plotted in figure 1.7, again a break in 1995 reflecting the addition of TVEs to the estimates. Emissions rose from 13.2 million tons (Mt) in 1985 to a peak of 19.5 Mt in 1995, equivalent to 23.7 Mt under the new definitions. It then fell and leveled off around 2000, before shooting up again to 21.59 Mt in 2003 (SEPA 1986–2004). This trend follows that of coal consumption in figure 1.2 reasonably closely—more than 90% of SO₂ emissions are from coal combustion—with only a small reduction in emissions per ton of

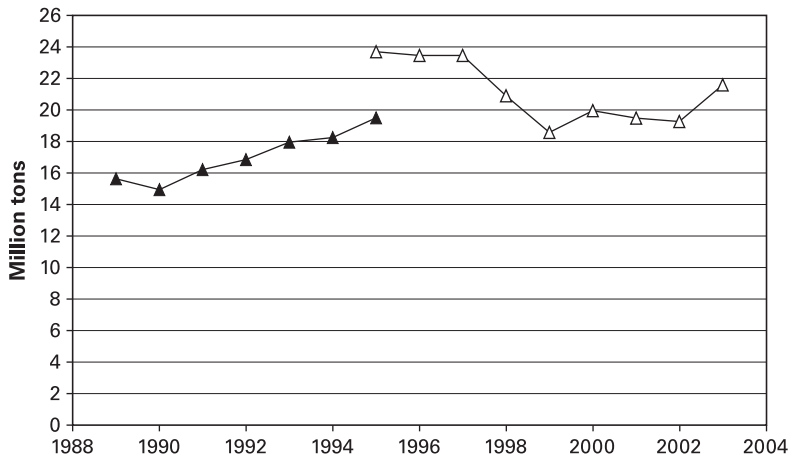


Figure 1.7

Emissions of sulfur dioxide in China. Data coverage extended in 1995 to include township and village enterprises. Source: SEPA 1986–2004.

coal. Official emission estimates are based on energy data, and such “bottom-up” inventories may thus be subject to some of the same uncertainties and errors discussed in section 1.3 on energy use.

Streets et al. 2000 conducted an independent assessment of SO₂ emissions through 1997 that was based on extrapolation of a detailed emission inventory developed for the RAINS-Asia project (Regional Air Pollution INformation and Simulation-Asia), which used energy-use data from the International Energy Agency. A follow-up inventory, adjusting official Chinese figures with omitted emission sources, estimated a total for 2000 (Streets et al. 2003). These figures are consistently higher than official data but show a comparable trend: from 17.90 Mt in 1985 to a peak of 26.21 Mt in 1996 (Streets et al. 2000), followed by a decline to 20.38 Mt in 2000 (Streets et al. 2003). An improved and updated inventory covering the more recent increases in SO₂ emissions may be forthcoming from this research group.

Nevertheless, conducting inventories of pollutant emissions in China is challenging work and uncertainties, while narrowing, remain large. Zhang (2005) included a review of the Chinese and international literature and plotted widely varying estimates of total annual SO₂ (and NO_x) emissions in China, from 1984–2004. It is noteworthy that for each year, the official estimates of SO₂ emissions are lower than all estimates by independent research teams. The emission uncertainties may

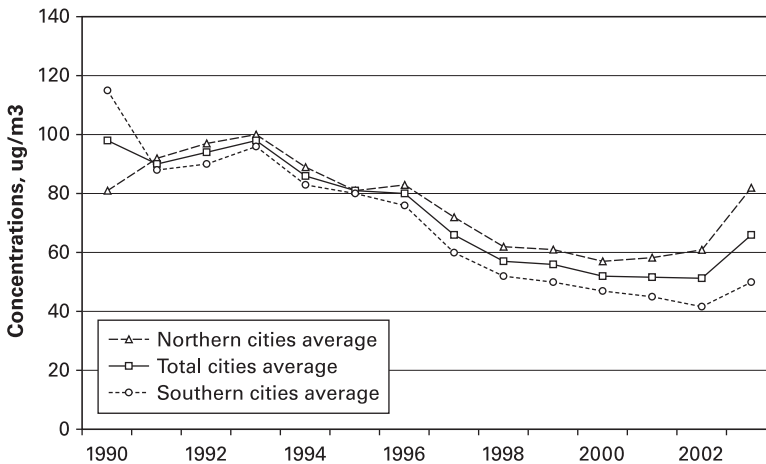


Figure 1.8

Average ambient sulfur dioxide concentration in cities of China. Source: Sinton et al. 2004, SEPA 2004.

soon be reduced further by new lines of evidence derived from satellite-based observations of SO_2 levels over China (Richter, Wittrock, and Burrows 2006).

Figure 1.8 gives average annual concentrations of ambient SO_2 reported for Chinese cities, similarly differentiated by SEPA into southern and northern cities. The concentrations steadily decreased from 1992 to 2000, leveling at a national average of $51 \mu\text{g}/\text{m}^3$ in 2002 before shooting up to $66 \mu\text{g}/\text{m}^3$ in 2003. Through much of the 1990s there was less geographical variance compared to TSP concentrations, with a steady pace of decline that is similar across regions. In recent years, however, SO_2 concentrations rose, an increase that preceded the rise in TSP concentrations, and the rise was greater in the north than the south. More than TSP, furthermore, the earlier progress in SO_2 concentrations in many major Chinese cities has now clearly and sharply reversed. Forty-two percent of cities did not attain the class II standard of $60 \mu\text{g}/\text{m}^3$ in 2003, compared to 29% in 1999 (SEPA 2004; PRCEE 2001). The annual average guideline of WHO is $50 \mu\text{g}/\text{m}^3$ (WHO 2002, p. 196).

The decline in SO_2 concentrations through the 1990s is attributed chiefly to the changing fuel structure of household energy use, in particular switching from coal combustion to natural gas, washed coal, liquid petroleum gas, and electricity (PRCEE et al. 2001). This took place despite the rise in aggregate emissions of the early 1990s shown in figure 1.7. These sources tend to be at ground level, directly

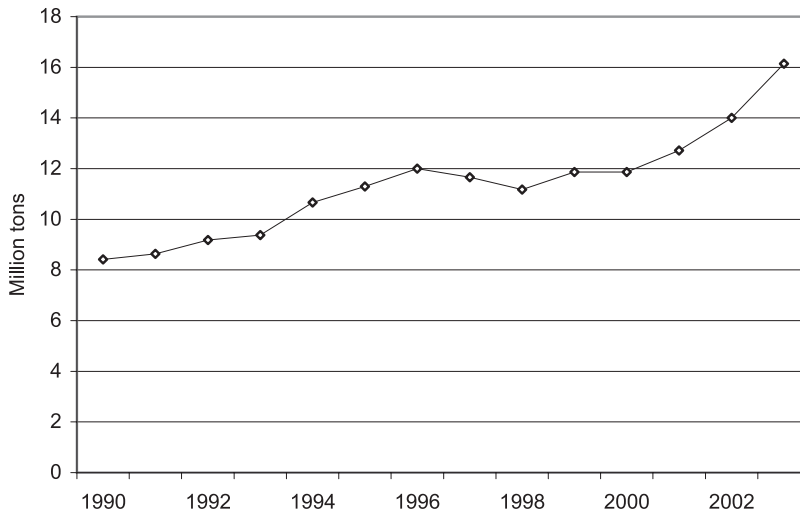


Figure 1.9

Emissions of nitrogen oxides in China. Source: Hao et al. 2002 for data through 1998; 1999–2003 data from personal communication with Hao research group at Tsinghua University.

affecting concentration measurements. Successful pollution control in urban areas may have been offset by growing emissions in nonurban sources and by higher stacks for urban sources allowing greater dispersion away from the cities. More recently, however, the trends of emissions and urban concentrations appear to have recoupled, the rise in SO_2 concentrations following rising emissions from higher coal use.

1.4.3 Nitrogen Oxides

NO_x emissions are associated with health damages, but they likely pose more health risk in their contribution to secondary chemical formation of ozone and fine particles. This volume considers NO_x only briefly, in chapter 5. There is no systematic accounting reported by the government of NO_x emissions, and we do not describe it in much detail here. Hao et al. (2002) estimated total anthropogenic emissions over two decades until 1998 on the basis of commercial energy consumption, NO_x emission factors, and fuel types, and the Hao research group has continued to calculate emissions in subsequent years. They estimate that total emissions increased from 4.76 Mt in 1980 to a peak of 12.03 in 1995, fell to 11.18 Mt in 1998, and rose again to 16.14 Mt by 2003, which reflects the coal and total energy trends described above (and which are the basis for the inventory). These trends are depicted in figure 1.9.

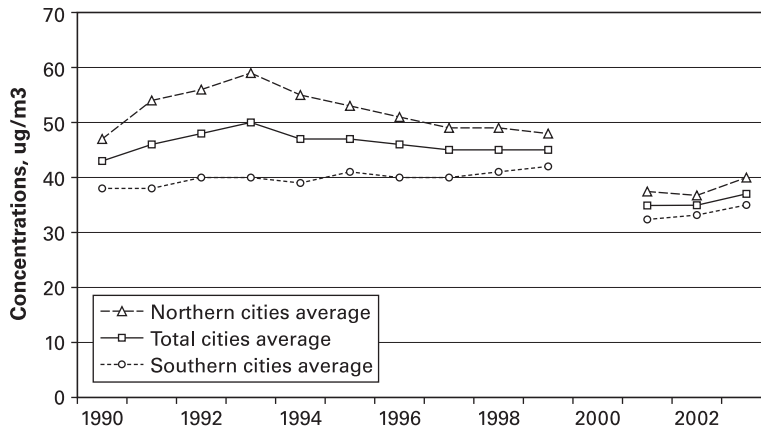


Figure 1.10

Average ambient concentrations of nitrogen oxides in cities of China. The measurements were switched from all nitrogen oxides (NO_x) to just nitrogen dioxide (NO_2) in 2001. Source: Sinton et al. 2004; SEPA 2004.

Hao et al. also analyzed source contributions of NO_x , in 1998, with industry estimated to have contributed 41% (4.59 Mt) of emissions, electric power 38% (4.23 Mt), and transportation 13% (1.45 Mt). Streets et al. 2003 provided an estimate for 2000 of 11.34 Mt, including emissions from biofuel and biomass combustion not included in the Hao et al. 2002 inventory.

Measured ambient concentrations reported by the government suggest that, when taken alone as a primary pollutant, NO_x has not been a serious risk in most Chinese cities. The average annual urban concentrations compiled by SEPA for all major cities did not exceed $50 \mu\text{g}/\text{m}^3$ in any year from 1990 to 1999, as shown in figure 1.10. The break in the graph in 2001 was when measurements were switched from NO_x to its major constituent species, NO_2 . These have been within the WHO guidelines for annual average NO_2 , of $40 \mu\text{g}/\text{m}^3$ (WHO 2002, p. 179). Until 2002 the official data showed offsetting trends of NO_x or NO_2 nationally, a steady downward path of concentrations in northern cities and an upward one in southern cities. PRCEE et al. 2001 noted that the trend also was better for smaller than larger cities and attributed this to rising emissions from the rapidly expanding vehicle populations in larger, southern cities with the swiftest economic growth.

This suggests that viewing the NO_x trends from a national perspective masks what is likely a swiftly worsening problem in a number of major urban areas with growing vehicle stocks and congested road networks. Recent official data shown in

figure 1.10 indicate that the long decline of NO_x in northern cities has reversed, whereas gradual growth of concentrations in southern ones continues. A new line of independent evidence gained a high profile in 2005 when satellite measurements by the European Space Agency reported in *Nature* indicated that, by 2004, an urban and industrial region of China stretching from Beijing to Shanghai had the worst tropospheric NO_2 levels in the world (Richter et al. 2005).⁷ Annual average levels in this region had risen around 50% since 1996, driven by especially large and accelerating increases in winter months. The Richter et al. estimates are of NO_2 in the lower atmosphere, not just at the surface, but they reflect the same emission sources and provide independent, scientific evidence of worrisome trends that official emission inventories and government-sited and -operated monitoring stations may not be picking up.

In summary, the emissions and urban concentrations of our target pollutants show, on balance, considerable progress in control through the 1990s but leveling or negative trends more recently. TSP emissions and ambient urban concentrations declined steadily and substantially before stabilizing in the current decade. SO_2 emissions rose and fell through the 1990s but have been rising again, whereas ambient urban concentrations steadily declined, leveled, and then by 2003 were sharply rising again. In many cities concentrations of these two pollutants exceed air-quality standards, especially TSP. NO_x emissions are not officially reported but, after some decline in the late-1990s, are likely rising again, and, after a long period of stability, ambient urban concentrations are also increasing, possibly dramatically.

1.5 Health Damages of Air Pollution in China and Research Framework

We are concerned about China's degraded air quality because it causes damages. The research of this volume focuses on health effects, the type of damage that most researchers believe dominate total pollution impacts, and the implications for the economy of policies to protect health. (Other studies of China have incorporated some non-health damages of pollution, including Hirschberg et al. 2003, O'Connor et al. 2003, and Wang and Smith 1999. We aim to consider nonhealth damages in future research.)

The burning of fossil fuels that cause much of this local air pollution also emits greenhouse gases, and these are widely expected to produce varied and pervasive global damages (IPCC 2001). Quantifying these effects is very difficult at this stage, and we do not attempt such estimates in this book. We do, however, discuss the effect of pollution control policies on the quantity of greenhouse gas emissions.

We now introduce our assessment framework, the details of each part given in separate chapters. Our research strategy follows what is commonly referred to as a pollution causal chain:

Component	Description	Chapter
1	From economic activity and energy use to emissions	1, 9
2	From emissions to concentrations (atmospheric transport)	4–7
3	From concentrations to human exposure	4–7
4	From exposures to health impact (dose-response)	4
5	Economic valuation of health impacts	8
6	National damage assessment by sector	9
7	Benefit-cost analysis of policies to reduce emissions	10

Our research (represented graphically in figure 1.11), makes primary contributions to components 1–3 and 5–7. Although we could not contribute here to component 4—air pollution epidemiology, which requires costly and time-consuming field

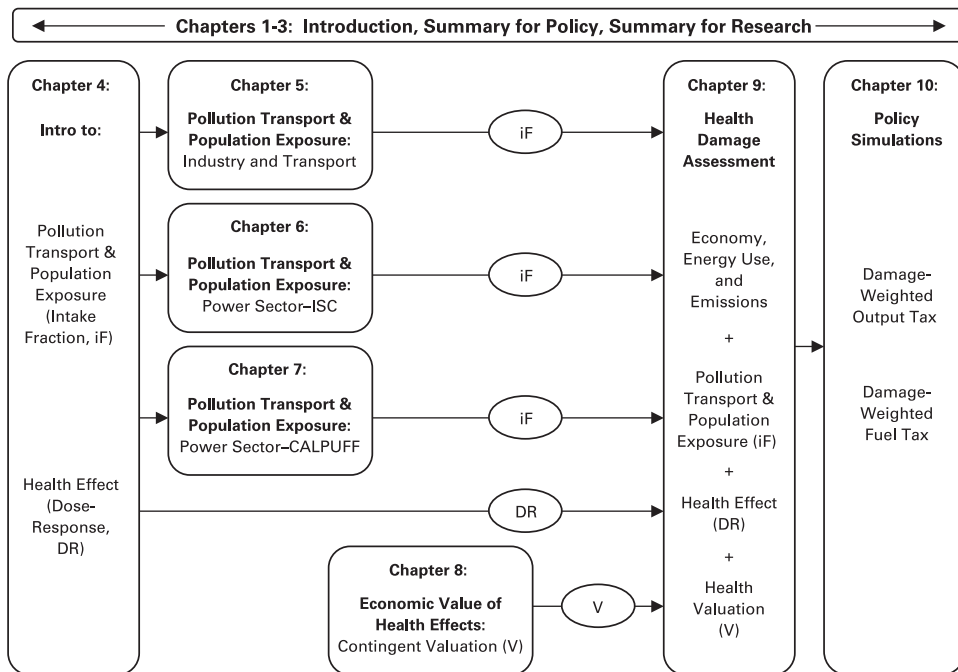


Figure 1.11
Research framework.

survey work—we do introduce the issues of defining and estimating functions that describe the human health response to pollution exposures. These are drawn from studies that have already been published. Components 1, 6, and 7 are implemented by using an economic model of China that identifies thirty-three distinct production sectors and one nonproduction sector (households), and traces the evolution of the economy over time. Earlier versions of this model have been used in previous studies of the effects of controlling carbon emissions and of the local health effects of controlling global pollutants (Garbaccio, Ho, and Jorgenson 1999a, 2000). This model generates the output of each sector for each period and with it the use of fossil fuels. The combustion of fuels and the production of goods generate emissions of PM, SO₂, and CO₂, and these are estimated using time-varying emission factors. This is a fairly well known procedure, as we describe in detail as part of the national analyses of chapter 9.

Components 2 and 3 are the focus of chapters 4–7. The requirement here is to translate emissions from a range of economic sectors into human exposures. Past assessments of this sort have typically adopted a damage function framework. In this approach, the emissions of the relevant pollutants from *all* sources are first estimated, along with characteristics of the source (e.g., stack height and diameter, pollutant exit temperature, or velocity). Researchers use atmospheric models to estimate how these emissions influence ambient concentrations or pollutant deposition at a number of locations. These typically incorporate detailed meteorological characteristics, sometimes allowing for chemical interactions among pollutants. Population distributions and locations are then used to determine at-risk populations. Finally, in a step summarized by component 4, concentration-response functions from epidemiological studies are combined with the exposure assessment to estimate population health risk.

Although this methodology is well established (ORNL and RFF 1994; EC 1995; Rowe et al. 1995; U.S. EPA 1999; Levy et al. 1999), it requires a large amount of very detailed information that is currently unavailable for a national-scale analysis in China.

Our aim therefore is to develop a framework to estimate health impacts on the basis of fundamental physicochemical properties but without all of the detailed input data that are now lacking. We wish to make a reasonable approximation of nationwide health impacts with a substantial, but not overwhelming, data-collection effort. By adopting a modular approach, we can identify methodological improvements to be tackled in turn. Additional refinements can be added as more data become available and more advanced atmospheric models are developed in China.

To make this approximation, we apply the concept of “intake fraction,” as developed by researchers at the Harvard School of Public Health and a number of colleagues at other universities and organizations. Intake fraction will be formally defined and explained in the chapter 4, but its basic meaning is the amount of material released from a source that is eventually ingested or inhaled. The concept has been developed for precisely the problem at hand: to conduct reasonable, if not precise, risk assessments with limited input information, as is often the only option in developing countries. Once the intake fraction of a source type has been estimated, population health risk can be simply quantified as the product of intake fraction, emission rate, and the unit health risk of the pollutant. The key research challenge is to determine reasonable intake fractions for different pollutants and source types.

Chapter 5 estimates intake fractions for four highly polluting sectors: iron and steel, chemicals, cement, and transportation. National inventories were not available, so our team collected detailed data from five cities across China. These data are supplemented by sample information for sources in the entire country. In assessing exposure for these sectors, we employed an air-dispersion model of relatively simple (Gaussian) form, ISCLT (Industrial Source Complex Long Term), which could be applied to the many sources of our database without overwhelming time and effort.

Chapter 6 estimates intake fractions from the electric power industry, which is by far the biggest user of coal. This uses a very large national emission inventory compiled by our team, chiefly from environmental impact assessment reports. This chapter also made use of the air-dispersion model ISCLT.

Chapter 7, like chapter 6, considers intake fraction and health risk in the power sector. This research was initiated to test the feasibility of applying a more advanced air-dispersion model, CALPUFF, that could cover a larger spatial domain and include secondary chemistry. The time and computational requirements to apply this model to our entire power plant database proved imposing, so we relied chiefly on the ISCLT results for the national power sector exposure assessment. The CALPUFF analysis, however, provides other information for the purposes of this volume (and stands also as a valuable independent study). It allows us to incorporate approximation of the health risks due to long-range air transport and secondary particles that the model of chapters 5 and 6 does not estimate. It also allows comparison to similarly conducted analyses in the United States, showing how higher population densities and more urban plant sites in China raise health risks for the same quantity of emissions.

With estimates of human exposure to air pollution for the five sectors, the next step is component 4, to apply dose-response functions from epidemiological studies

to estimate health effects. As mentioned above, we do not offer new air pollution epidemiology, which requires expansive and costly field research. It instead relies on results in the published literature. A number of such studies were conducted by colleagues of our team with support of earlier stages of our program (Xu 1998; Wang et al. 1999; Xu et al. 2000; Venners et al. 2001, 2003).

Attributing the incremental morbidities and mortalities (e.g., the cases of actual respiratory disease and related death) to pollution emissions from particular industries provide a powerful quantified case for pollution control. Many policy makers, including those in China, also like to see such health impacts translated into economic terms, to help compare costs and benefits of different policy interventions.

The monetization of health damages—component 5—is acknowledged even by its advocates as an imprecise science. Recognizing the uncertainties, we nevertheless devote considerable effort to it here because it offers useful information to help communicate and prioritize environmental risk in real-world policy processes.

A number of estimation methods, with associated terminologies, have been developed to value damages to health. These include assessments of “willingness to pay” (WTP), determined by “revealed preference” observed from actual market transactions, or by interview-based surveys (“contingent valuation,” or CV). They also include a more conservative approach favored by many Chinese researchers, the “human capital” method that values health outcomes chiefly by their effect on expected earnings. The assessment includes a new contribution to this field in chapter 8, a CV study, and a brief overview of valuation methods and underlying theory is included in that chapter for interested readers.

To bring all of the foregoing together and assess national health and economic damages on a sector basis—component 6—chapter 9 begins with estimates of national emissions of TSP and SO₂ for each sector. This is done in our environmental-health submodel, as described in that chapter. It then brings the intake fractions of chapters 5–7 and the dose-response coefficients recommended in chapter 4 together to estimate the health effects. Our methodology draws on World Bank (1997) and examines the impact of pollution on eleven health effects, the two most important being premature mortality and chronic bronchitis.

Because of uncertainties in methodological approaches to valuation, chapter 9 applies a range of estimates from the literature, including those of chapter 8 and values derived in studies from other countries and scaled to Chinese incomes. The result is valuation of the health damage from air pollution in each sector. This allows us to derive the marginal damage per unit output for each sector as well as the marginal damage per unit fuel combusted. The estimated total damages for each

sector and their distinct marginal damages help us rank sector priorities in pollution reduction.

The estimated marginal damages indicate the relative importance of each unit of output and fuel use to the total air pollution problem. We use these measures to analyze policies that tax output and fuels in proportion to the damage caused (a “green tax,” or, in economic jargon, a Pigovian tax). This policy analysis is reported in chapter 10 and is accomplished by use of the economic model combined with the environment-health submodel. With the integrated model, we are able to analyze both the benefits and costs of control policies of component 7. The benefits have the form of reduced health damage, and the costs are in terms of lower consumption and GDP.

We emphasize again that this research is an incremental advance, not a definitive one, in the development of a framework to assess health damages of air pollution and associated policies in China. Its design is such that future refinements in any component may be immediately used in subsequent applications to improve the policy analysis. As such, we note now that it suffers a number of gaps. The first caveat is that our analysis has a limited treatment of secondary pollutants, using a simple approximation method based on results of chapter 7 to estimate exposures to secondary particulates, and leaving ozone to future research.

Second, we carefully investigate exposures in five major sector sources of ambient air pollution but use simpler approximations for the others. Because of several factors, including the scale of energy consumption in these other sectors, they are likely to have substantially lesser effects on air pollution than the ones modeled, save one: the residential sector. Unlike industrial and transportation sources, households have complex emission and exposure characteristics and thus pose an analytical challenge meriting a separate assessment that could easily be as large and challenging as the one reported here. It is essential to spotlight indoor air pollution, particularly in the rural residential sector, which could be one of largest sources of health damages of air pollution in China.⁸ Very different sorts of policies, however, are generally needed to target pollution control in the residential versus industrial and transport sectors.

This is one of several reasons we do not believe it is accurate or even especially useful in a policy sense to reduce estimates of health damage from air pollution to a single number, the percentage loss to GDP, as is often reported and emphasized in the literature. Rather than estimating aggregate national cost, we are more interested in—and such models are better suited for—differentiating the relative health and economic impacts of emissions from the many sectors and examining the

impacts of prospective policy interventions on emissions, public health, and growth of the economy at large.

1.6 Review of Other Assessments of Health Damages of Air Pollution in China

A substantial literature estimating the costs of environmental degradation in China has developed in the past fifteen years, varying widely in ambition. In light of the focus of the current study on advancing methods of estimating the health damages of air pollution—which typically dominate costs of environmental degradation more generally—we spotlight studies of these damages, especially international collaborations with Chinese research institutes employing methods from the worldwide literature. The brief descriptions of the different contributions here are intended as a reference for interested readers and permit us to credit prior efforts and contrast several with our assessment. We do not provide a comprehensive discussion or critique of the methods and results of each.

Table 1.2 summarizes key features of these studies. We list a number of representative characteristics across the different analyses but caution against direct comparisons, because each study differs in objectives, methods, scope, base year, and assumptions. In addition, many key elements are not included in the table (e.g., health damages from SO_2 and morbidity). These are each complex research efforts, and the reader is directed to the original sources for a fair understanding.

We emphasize a key modeling limitation inherent in the following summary. To some degree, researchers in this field, as in many others, must trade off two interests: high resolution in analysis of the energy-to-exposure pathway versus broader approaches that include regional air dispersion and take into account economic feedbacks, so-called general equilibrium effects.

The high resolution of a “bottom-up” assessment allows a detailed analysis, for example, of specific power-generation technologies or of all emission sources in a city. Environmental health scientists increasingly emphasize that the health impacts of pollution exposure depend critically on not just small spatial scales, but also small temporal ones, such as the timing of daily activities. Such assessments require data resolved at fine grids, such as emissions at the plant level and population by small districts.

To date, a number of researchers have conducted bottom-up assessments to assess exposure and health in China on subnational geographical scales. The studies listed below include several conducted for municipalities or provinces from the bottom up. Although there are also large approximations in these models, and no research team

Table 1.2
International collaborations on health damage of air pollution in China

Name (Year)	Geo- graphical Domain	Sectors	Air Pollutants	Types of Air Pollution Damages	Mortality PM Dose-Response Coefficient	Method(s) and Central Value of Statistical Life or Year of Life Lost	Central Estimate of Aggregate Effects
National studies							
World Bank (1997)	China	All, based on measured ambient concentrations	PM ₁₀ , SO ₂ , Pb	Mortality and morbidity (urban ambient and rural indoor air); also acid rain on crops, materials, ecosystems	6 deaths per 1 million people per µg/m ³ of PM ₁₀	U.S. WTP scaled to China by GDP ratio: \$31,762/life (rural), 60,000/ life (urban); human capital method: \$4,748/life (rural), 9,970/ life (urban)	178,000 annual urban prema- ture deaths from exceeding ambient stan- dards, 111,000 from rural indoor air levels; equiva- lent to 7.1% of GDP in early 1990s using WTP method
ECON (2000)	China	Not applica- ble; study develops a model but does not apply it to data	PM ₁₀ or SO ₂ , O ₃ , Pb	Mortality and morbidity (ambient and indoor air); also acid rain and ozone on crops, materials	0.4 (0.0, 0.6) percent increase mortal- ity per µg/m ³ of PM ₁₀ ; equivalent to 24 deaths per 1 million	100× per capita GDP (based on Western ratios): \$77,000/life	Not applicable; study develops a model but does not apply it to data

O'Connor et al. (2003)	China, based on Guangdong	All, 62-sector CGE economic model	PM, SO ₂ , NO _x , O ₃ , VOCs, CO ₂	Mortality and morbidity (ambient air). Also O ₃ on crops	2.2 (0, 4.1) deaths per 1 million adults per µg/m ³ of PM ₁₀ ; 0.7 for infants	Taiwan WTP scaled to China by GDP ratio: \$43,275/life (for all China), \$68,973/life (for Guangdong only).	Net benefits of 0.05% of output due to a 10% carbon reduction
Hirschberg et al. (2003)	China, based on Shandong	All, based on electric power	PM, SO ₂ , NO _x , NH ₄	Mortality and morbidity (ambient air). Also acid rain on crops, climate	1.57E-4 YOLL per year-person-µg/m ³ of PM ₁₀	Western WTP scaled to China by PPP GNP: \$430,000/life, \$15,710/YOLL	1 million premature deaths annually from all air pollution; 9.1 million YOLL per year: equivalent to 6–7% GNP
Sector studies							
Wang and Smith (1999)	China	Electric power, households	PM ₁₀ (including SO ₄ ⁻²), CO ₂	Mortality and morbidity (ambient and indoor air)	0.1 (0.04, 0.3) percent increase mortality per µg/m ³ of PM ₁₀	24,000× average daily wage (based on U.S. ratio), adjusted by PPP: \$123,000/life	75,400–122,500 annual premature deaths avoided from BAU in 2010, depending on scenario

Table 1.2
(continued)

Name (Year)	Geo- graphical Domain	Sectors	Air Pollutants	Types of Air Pollution Damages	Mortality PM Dose-Response Coefficient	Method(s) and Central Value of Statistical Life or Year of Life Lost	Central Estimate of Aggregate Effects
Feng (1999)	China, Korea, Japan	Electric power	TSP, PM _{2.5} , SO ₂ , SO ₄ ⁻² , NO ₂ , NO ₃ ⁻ , CO, CO ₂ , CH ₄ , N ₂ O	Mortality and morbidity (ambient air); also multiple pollutants on crops forests, ecosystems, materials, visibility, climate	Multiple coefficients, for both chronic and acute mortality, and for different ages	U.S. WTP scaled to China by PPP GNP: \$339,500/life	Not applicable; study estimates damages from a hypothetical power plant
Provincial study							
Aunan et al. (2004)	Shanxi	Industry, electric power, coking, households	PM ₁₀ , SO ₂ , CO ₂	Mortality and morbidity (ambient air)	2.2 (0, 4.1) deaths per 1 million adults per µg/m ³ of PM ₁₀ ; 1.2 (0.8, 1.7) for infants	100× per capita local GDP (based on Western ratios): \$63,000/life	Not applicable, study estimates avoided deaths and savings of six technology options

City studies						
ECON et al. (2000)	Guangzhou	All sources in urban domain	PM ₁₀ , SO ₂ , NO _x , CO ₂	Mortality and morbidity (ambient air); also acid rain on materials	2.2 (0, 4.1) deaths per 1 million adults per µg/m ³ of PM ₁₀ ; 0.7 (0.4, 0.9) for infants	Taiwan WTP scaled to China by GDP ratio: 620,288 yuan/ life 1,420 prema- ture adult deaths an- nually from all air pollu- tion; 110 pre- mature infant deaths annually
Lvovsky et al. (2000)	Shanghai	Large indus- try, small industry, electric power, households, urban transportation	PM ₁₀ , SO ₂ , NO _x , CO ₂	Mortality and morbidity (ambient air); also reduced visibility, soiling, materials, and climate	0.194 percent increase mortality per µg/m ³ of PM ₁₀	U.S. WTP scaled to China by income ratio: \$72,859/ life 3,979 prema- ture deaths annually from air pollution; equivalent to 5.5% local income
SAES et al. (2002)	Shanghai	All sources in urban domain	PM ₁₀ , SO ₂ , NO _x , CO ₂	Mortality and morbidity (ambient air)	0.43 (0.26, 0.61) percent increase mortality per µg/m ³ of PM ₁₀ (long-term); 0.028 (0.01, 0.046) percent (short-term)	WTP by CV for Chongqing, income adjusted to Shanghai: US \$108,500/life 647–5,472 avoided deaths in 2010 depending on energy scenario; 1,265–11,130 in 2020
Kan and Chen (2004)	Shanghai	All sources in urban domain; based on measured ambient concentrations	PM ₁₀	Mortality and morbidity (ambient air)	0.43 (0.26, 0.61) percent increase mortality per µg/m ³ of PM ₁₀	WTP by CV for Chongqing, income adjusted to Shanghai: US \$108,500/life 4,780 prema- ture deaths annually from air pollution; equivalent to 1.0% local GDP

Table 1.2
(continued)

Name (Year)	Geo- graphical Domain	Sectors	Air Pollutants	Types of Air Pollution Damages	Mortality PM Dose-Response Coefficient	Method(s) and Central Value of Statistical Life or Year of Life Lost	Central Estimate of Aggregate Effects
Wang and Mauzerall (2006)	Zaozhuang, and region	All sources in urban domain (biogenic in addition to anthro- pogenic)	PM _{2.5} , PM ₁₀ (including SO ₄ ⁻² , NO ₃ ⁻), SO ₂ , NO _x , CO, NH ₃ , VOCs	Mortality and morbidity (ambient air)	0.58 (0.2, 1.0) percent increase mortality per µg/m ³ of PM _{2.5} ; 4.7E-4 YOLL per year-person-µg/ m ³ of PM _{2.5}	WTP by CV for Chongqing, inflation adjusted: US \$34,235/life	6,000 prema- ture deaths annually from air pollution; equivalent to 10% of local GDP
Peng et al. (2002)	Shijiazhuang	All sources in urban domain	PM ₁₀ (including SO ₄ ⁻²)	Mortality and morbidity (ambient air)	6 deaths per 1 million people per µg/m ³ of PM ₁₀ ; 0.08 percent increase mortality per µg/m ³ of PM ₁₀	WTP by CV: \$160,000/life	251 annual premature deaths from exceeding ambient stan- dards; equiva- lent to 4.3% of local GDP

Notes: Base years: We are unfortunately unable to list all of the base years of these assessments, required for careful comparisons across studies. The base years often differ by study component (e.g., emissions vs. valuation) and in many cases are unclear because they are not described clearly or at all in published reports. We reemphasize that this table is merely introductory and researchers should consult the original literature. General abbreviations: CGE, computable general equilibrium (economic model); GDP, gross domestic product; GNP, gross national product; PPP, purchasing power parity; µg/m³, micrograms per cubic meter; YOLL, year of life lost; WTB, willingness-to-pay; CV, contingent valuation; and BAU, business as usual. Pollutant abbreviations: PM, particulate matter; and PM₁₀ and PM_{2.5}, categories of PM under aerodynamic diameters of 10 and 2.5 microns; TSP, total suspended particulate matter; SO₂, sulfur dioxide; SO₄⁻², sulfate particles (of various chemical forms); NO_x, nitrogen oxides; NO₃⁻, nitrate particles (of various forms); O₃, ozone; CO, carbon monoxide; Pb, lead; VOCs, volatile organic compounds; CO₂, carbon dioxide; CH₄, methane; and N₂O, nitrous oxide.

has modeled literally every single source of pollution in a jurisdiction, such studies can provide valuable guidance for local decision making.

There is also a pressing need, however, to understand effects at the regional or national level, especially for a full economic analysis. Some pollutants such as SO₂ travel hundreds of kilometers, and a comprehensive accounting of health damages requires a regional scope. More important, a consideration of only the direct local or industry-specific effects of damages and policies may miss out a larger total effect on the rest of the economy. For example, a policy that affects the electric power industry will change the price of electricity. This may affect the cost of making steel and cement needed to construct power plants, which in turn may lead to further changes in the demand for electric power. In general such policies would have ramifications across the entire economy today and, by influencing the rate of investment, would affect the economy in the future. A comprehensive benefit-cost assessment of national pollution control policy needs to consider not just the impact of policies on directly affected enterprises and sectors, but also the potentially large indirect effects such interventions may have throughout the economy both now and in the future.

We must recognize that localities in some circumstances can set their own pollution-control policies, but the Chinese government must also make environmental policy choices on a national basis. Without a truly massive data-gathering and research effort, it is impossible to conduct a bottom-up modeling assessment for the entirety of China, even if one were to accept the assumptions made by the teams below in studies of more limited domains. In practice, the setting of national environmental policies can be analyzed only using more abstract “top-down” models that take the national damages and economic interactions into account. It is to advance the very limited body of this type of analysis in China that this volume, structured around a national economic model with general equilibrium features, was conceived and conducted.

1.6.1 National Studies

China Research Academy of Environmental Sciences (CRAES 1999) The estimation of environmental losses in China can be traced to the end of the 1980s. Most of the first studies were modest in scale and scope, usually conducted for a geographical region or city. A report by CRAES (1999) lists a large number of these studies, some quite informal and all in Chinese. For instance, Guo and Zhang (1990) conducted one of the first national assessments of the losses of environmental and ecological damages. By use of human capital valuation methods and other

approaches, they estimated an average annual loss of 38 billion yuan from 1981 to 1985, about 6.75% of GDP in 1983 (CRAES 1999). Air pollution was estimated at an average of 12.4 billion yuan, or 2.19% of GDP.

World Bank (1997) The first large-scale collaboration to apply methods common in international research to assess health damages of air (and water) pollution was led by the World Bank (1997). *Clear Water, Blue Skies* was a valuable initial effort to estimate impacts on a nationwide basis. Faced with data constraints but unable to conduct major field efforts, the team employed approximation methods that included borrowing key relationships in the emission-to-exposure pathway from analogous studies in Eastern Europe. It is impossible to quickly summarize the study's results, but for example it estimated 178,000 premature deaths per year from exceeding China's urban air-quality standards.

The World Bank (1997) employed two approaches to valuation of health effects. The first was based on "revealed preference" studies in the United States, transferred to China by linearly scaling to the ratio of GDP per capita. This yielded an estimated unit value of mortality in urban China of US\$60,000, one factor in a subsequent estimate that air pollution costs China nearly US\$50 billion per year, or 7.1% of GDP.⁹ This result was controversial in China, where, among other reservations, experts generally preferred a second, more conservative, "human capital" form of health valuation. This alternative approach reduced the loss estimate to \$20 billion per year, or 2.9% of GDP.

The World Bank study was an important first assessment and was updated in previous work of members of the current project (Ho, Jorgenson, and Di 2002, see below). As mentioned earlier, the effort of this volume was initiated in part to improve more extensively on World Bank 1997, with a more detailed analytical approach that is also rooted in more data gathered from fieldwork in China. To enhance the applicability of results, the assessment here is also designed to attribute damages differentially to emissions and concentrations modeled from key economic sectors. This allows a national-level damage attribution and sector prioritization that the World Bank approach cannot provide. We based our decision to focus initially on health impacts of PM and SO₂ in part on the overwhelming dominance of these damages over those of other environmental pathways (e.g., agricultural impacts and acid rain) in World Bank 1997.

ECON Centre for Economic Analysis (ECON 2000) The World Bank and SEPA subsequently initiated a program to test and revise the methods and estimates of

World Bank 1997, and eventually develop a common framework for annual damage assessment in China, on national and even provincial scales. Titled the “China Environmental Cost Model,” it intends to assess a wide range of both air- and water-pollution costs. This encompassing initiative was nearing completion at the time of writing, and it cannot be cited or detailed here. One of the initial undertakings, however, was to commission consultants to develop a modeling framework that includes impacts on human health, materials, and agricultural production. This is described in the ECON 2000 report.

The ECON 2000 model framework focuses on PM_{10} and SO_2 , taking into account their interrelationships, and also adds independent impacts of ozone and lead. For valuation, it reviews the worldwide literature and suggests a convenient formula that values a statistical life at 100 times the per capita annual GDP. The purpose of the study was to develop the model, not to apply it, and thus it did not yet attempt to generate national damage estimates in its report. Its comparison to World Bank 1997 is limited to constituent elements of the assessments, such as the number of morbidity cases per 1 million people per $1 \mu g/m^3$ increase in ambient PM_{10} .

By applying directly the ratio of GDP per capita between China and the United States, the ECON model converted the unit economic loss from respiratory illness in the United States to the unit economic loss from the same illness in China. However, the calculated annual medical cost is three times higher than the actual costs in Beijing. Thus, a study of Tsinghua University et al. (2002) modified the model by taking one third of the unit economic loss from morbidity or death in the model as the unit economic loss from respiratory illness in Beijing.

A primary strength of the pending China Environmental Cost Model, based on ECON 2000, should be its breadth, assessing water quality in addition to air quality, and including lesser damages from additional pollutants and nonhealth pathways. It is also designed for relatively easy and routine application. The assessment of this volume differs in its underlying economic structure, distinguishing health damages by sector and integrating them directly into a model with general equilibrium features. This allows it to consider full impacts throughout the economy in assessing the costs and benefits of given policies. An additional difference is its more extensive consideration of population exposures.

Ho, Jorgenson, and Di 2002 This earlier analysis of health damages from local air pollution used information and estimates in World Bank 1997 and calculated national health damages caused by each industry. It examined the costs and benefits

of national green tax policies, the effects on both emissions and the economy. This study noted the very high damages estimated for the transportation sector, something that had not been greatly emphasized previously.

O'Connor et al. 2003 This is a co-benefits study using a two-region economic model (in the jargon, a computable general equilibrium model, or CGE; “co-benefits” refers to the association of both global and local environmental benefits of emission control). It considers air-dispersion characteristics in Guangdong province in relative detail and then extrapolates them to the rest of China. O'Connor et al. employed two dispersion models, both a simple “stack-height-differentiated” one as in Ho, Jorgenson, and Di 2002 for primary pollutants and health impacts and another for estimating formation of ozone, a secondary pollutant that damages crop productivity. It asked how much the welfare costs of an energy tax for 5–30% carbon reductions in 2010 from a baseline might be offset by benefits to health and agriculture.

One result of this study that is especially noteworthy for its divergence from conventional wisdom is its conclusion that pollution damages to agriculture (from ozone in particular) may outweigh pollution effects on health (from particulates and SO₂) in economic terms. It thus asserts that damage avoidance in agriculture provides a larger proportion of the ancillary benefits of emission reductions.

O'Connor et al.'s incorporation of the damage estimations in a CGE model to consider the economywide effects of emission-control policies makes it closest of all the studies to the integrated analysis in our program reported in chapter 10. It has an advantage in its consideration of agricultural impacts, whereas chapter 10 uses our team's more elaborate estimate of pollution dispersion and exposure.

Hirschberg et al. 2003 A study by a Swiss-Chinese-American team conducted a detailed cost assessment of ambient air pollution from the power sector in Shandong province, extended in unspecified manner to the economy of China. Capturing local features that the high resolution of its provincial scope would allow, this “full scope, bottom-up” effort applied the EcoSense model to Shandong power sector sources and scenarios into the future. This model packages atmospheric transport and chemical conversion, population-based exposures, dose-response relationships, and valuation models, the latter two adapted from industrialized countries.

Among Hirschberg et al.'s conclusions are that, extrapolated to all of China, roughly one million Chinese die prematurely each year of ambient air pollution from all sources, with costs corresponding to 6–7% of GDP. Health costs, particularly

mortality, dominate other damages, such as to crops. A unique contribution of this study is noting a strong influence of agriculturally produced ammonia on health damages, because of its central role in chemical transformation of SO₂ into sulfate particles. They conclude that the major part of damage to health is caused by these secondary pollutants. The ultimate focus of this study was estimating external costs of electric power for use in analyzing power-sector technology scenarios in the future.

1.6.2 Sector Studies

Wang and Smith 1999 These researchers conducted a national health damage analysis for two sectors, electric power and households. This was a co-benefits study, the aim of which was analysis of four energy scenarios for China to achieve 10% and 15% reductions of greenhouse gases by 2010 and 2020, respectively. They compared technology pathways to reach least-cost reductions in both global warming potential and unit dosages of SO₂ and particulates, which were then translated into changes in health risk. Finally, the team estimated marginal net economic costs of achieving GHG targets by alternative coal use in these sectors, calculated from valuation of health benefits and incremental costs of energy changes. Health valuation was adapted from U.S. figures, which were adjusted by relative wage levels and purchasing power in China.

The results spotlighted enormous averted health damages as a co-benefit of achieving a GHG target, particularly through fuel substitution in the household sector. For instance, central estimates of avoided premature deaths in 2020 of the three scenarios over business as usual ranged from 154,400 to 185,200, against a projected total mortality in that year of 14 million. The study was designed to compare health co-benefit levels of different ways to achieve GHG reduction, not to calculate aggregated health damages of air pollution.

Feng 1999 This study, a published Ph.D. dissertation, assessed a hypothetical, characteristic power plant, at two possible locations in China. It merits inclusion here for its ambitiously comprehensive scope. The study modeled emissions and concentrations of a wide range of primary and secondary pollutants, both chronic and acute mortality and morbidity effects, and additional impacts on crops, forests, ecosystems, materials and visibility and included some damages in neighboring countries. It also considered emission of several GHGs, and attributed costs of climate change based on estimates from global studies. For valuing health effects, it

translated results of WTP studies from other countries and adjusted them for population age groups. Feng calculated the total environmental damages of the plant without pollution control and then evaluated the cost-effectiveness of current and prospective policies.

1.6.3 Provincial Study

Aunan et al. 2004 This project had a similar co-benefits objective and approach as Wang and Smith 1999 but applied to the single province of Shanxi and not to China as a whole. They posed a number of options to abate emissions of SO₂, particulates, and CO₂ in industry, power generation, and rural households. These included pre-combustion options of coal washing and coal briquetting, and combustion options of improved management, boiler replacements, cogeneration, and modified boiler designs.

Aunan et al. 2004 made conservative estimates of health impacts of these prospective interventions by focusing on eighteen cities in Shanxi for which data were available. Valuation of premature deaths avoided by adoption of each alternative used a rule of thumb of 100 times per capita local annual GDP, drawn from the analogous ratios resulting in Western studies. The study concluded that all six options are profitable in a socioeconomic sense. The least-cost avoidance of health damages, however, happened to be the most costly form of carbon abatement, which indicates a divergence of priorities depending on local or global environmental objectives.

1.6.4 City Studies

ECON et al. 2000 The aforementioned research by ECON 2000 was incorporated into “Guangzhou Air Quality Action Plan 2001,” which was conducted by Norwegian and Guangzhou researchers to advise a package of least-cost pollution control options that would meet SO₂, NO_x, and TSP targets already set by municipal officials. Assessment of a number of technology and policy options yielded cost curves for SO₂ and NO_x control that demonstrated the relative ease of meeting the former (along with TSP, which is assessed somewhat differently) versus the latter.

The study developed its own comprehensive emission inventory on the basis of a large survey and estimated gridded pollutant concentrations with a dispersion modeling package. It found encouraging correspondence with measured levels. Combined with population data, the resolution of the model allowed the team to distinguish exposure characteristics of different pollutants and sources. Modeling

health effects of given control options was then methodologically straightforward in such a comprehensive, “bottom-up” model. The study employed WTP valuation from research in Taiwan, scaled to China by per capita GDP.

Lvovsky et al. 2000 In this study, a World Bank-based team developed a “rapid assessment model” to estimate environmental damages and attribute them to various sources. It investigated six cities around the world, including Shanghai. The damages it assessed included health, reduction in visibility, soiling, material damages, and climate change.

The spreadsheet-based methodology of this study was purposely simple. Using a dispersion model that aggregated large sources in the center of a city and dispersed pollutants equally in all directions, for instance, the model was designed for quick use to generate a rough characterization of damages in data-limited contexts. Valuation of a statistical life was based on U.S. WTP scaled to China by income ratio. By use of these methods, health damages from air pollution in Shanghai totaled \$730 million in 1993 dollars, 5.5% of Shanghai’s income. Chiefly because of exposure characteristics, health damages were deemed greatest from small stoves and boilers in households and industry in Shanghai.

Shanghai Academy of Environmental Science (SAES) et al. 2002 Another study of Shanghai was supported by the China Council for International Cooperation on Environment and Development, SEPA, the Shanghai Environmental Protection Bureau, U.S. EPA, and others. It estimated health impacts of air pollution in 2010 and 2020, under a number of energy scenarios diverging from a base case. The study employed an atmospheric model originally designed to investigate acid rain that was adjusted to an urban scale. It estimated health effects of PM₁₀ exposures, including acute and chronic mortality, chronic bronchitis, and several others, by use of a mix of Chinese and Western epidemiology. SAES et al. monetized premature mortality using a simple CV study by Wang et al. (2001, from an early study of Chongqing by our Harvard University China Project, with multiple collaborators). For morbidity endpoints, the study applied income-adjusted values from the West or “cost-of-illness” estimates. The project then reported the total economic damage of its scenarios, but not as percentage of GDP. An associated study of Beijing has now taken place, but a full report was unavailable at the time of writing (Tsinghua University et al. 2005).

Kan and Chen 2004 Health scientists from Fudan University who participated in SAES et al. 2002 above followed with another assessment of Shanghai. Rather than

model emissions and air dispersion, they evaluated the effect of measured annual average daily PM_{10} for 2001 from concentrations observed at a background station. Applying the same concentration-response coefficients and valuations as the prior study, they attributed nearly 11% of annual adult deaths to the incremental PM_{10} . Premature mortality comprised nearly 83% of the estimated aggregate economic damage, 1% of Shanghai local GDP. This brief study concluded with a valuable discussion of uncertainties for consideration in future research.

Wang and Mauzerall 2006 This study examined impacts on health of particulate matter from sources in the city of Zaozhuang, Shandong. One of its primary strengths was its comparatively sophisticated treatment of atmospheric transport, chemical transformation, and deposition. After constructing an inventory of both anthropogenic and biogenic emissions, the study used the Community Multiscale Air-Quality Modeling System (CMAQ) to simulate ambient concentrations of primary and secondary particulates ($\text{PM}_{2.5}$) across a multiprovince domain. It investigated the regional health damages from Zaozhuang emissions both in 2000 and in 2020 under business-as-usual and two alternative technology scenarios. Unlike many studies that focus on acute mortality, this one included chronic effects. It calculated premature mortality by using two methods and then applied the Wang et al. 2001 CV results to value the damages. The result was an estimate of damages equivalent to 10% of Zaozhuang's GDP in 2000, a proportion growing to 16% by 2020 absent interventions like the modeled technology options.

Peng et al. 2002 This study examined health damages of sulfate particulates in the city of Shijiazhuang, by using an urban-scale, high-resolution, atmospheric puff model that featured secondary chemistry and took account of background concentrations. For valuation, it employed results of an unpublished CV study by Li, Schwartz, and Xu (1998). Peng et al. concluded that mortality and morbidity costs of sulfate amount to more than 4% of urban GDP in the year 2000, and their assessment of control options spotlighted the cost-effectiveness of adopting low-sulfur coal.

1.7 Conclusion

Two observations are in order to conclude this introduction and review. It is clear that China's growing economy and dependence on fossil fuels, especially coal, have engendered a number of serious air-quality problems and that these have substantial

impacts on human health. China has made considerable strides in addressing some of the most critical types of air pollution, as evidenced by declining measured concentrations of SO₂ and particulates in urban areas through the 1990s. Few living in China's cities through this decade would dispute this. This advance has undoubtedly benefited public health. It is partly testimony, however, to how severe air quality was to begin with, and conditions remain worse than official standards in many places. Moreover, despite the progress, the positive trends have recently leveled off (TSP) or reversed (SO₂ and NO_x), possibly sharply, as the economy continues to boom. Newer forms of pollution, such as photochemical smog, are also now becoming serious problems in a number of cities and their downwind regions.

What is also clear is that there is no shortage of interest in China's air pollution. Aside from public and official concern, a body of research is developing to better understand both the nature of the hazard itself and the damages that it can cause, to human health, to the local economy, and to the global environment. The research reported in this volume is one extensive effort to assess the costs of pollution and to incorporate the results in tools to inform policy, but it is one of many studies to date on this set of issues. The next chapter provides a brief summary of the results of our research collaborations for such policy inferences and sets the stage for the full reporting of our assessment in the second part of the book.

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Notes

1. Intensity measures include both narrow concepts like tons of coal per ton of steel and broader ones like total primary energy per yuan of GDP.
2. Coal transformed to coke is counted as industrial consumption.
3. Specifically, the balance terms—"coal available for consumption" minus "coal consumption"—have magnitudes of tens of millions of tons over 1990–1998 (33–42 Mt in 1990–1995, 61–75 Mt in 1996–1998; NBS 2001, table 4.5) and rise suddenly to hundreds of

millions of tons for 1999–2001 before dropping back to the previous scale in 2002–2003 (228 Mt, 264 Mt, 278 Mt, 70 Mt, and 58 Mt, respectively; NBS 2005, table 4.5). We note another indication of data problems for this period that our team happened upon. Adding up 1999 coal consumption listed *by province* in table 5.16 of NBS 2001 yields a total that exceeds by 165 million tons the oft-cited *national* coal consumption listed in table 4.5 of the same source. Such inconsistencies also occur in prior years, but on a much smaller scale.

4. As we will explain later, we do not consider severe indoor air pollution problems in rural China, which pose a major additional threat to public health. Considerable air pollution information is now conveniently available in the online version of the *State of the Environment in China*, published by SEPA, at www.zhb.cn.

5. We do this in chapter 4.

6. The term used for combustion emissions in the official books, *yanchen*, literally means “smoke dust,” whereas *fenchen* is the dust from production processes.

7. The *Nature* article was overinterpreted in some international media coverage, for instance suggesting it indicated that Beijing had become the “air pollution capital of the world” (Watts 2005). These largely ignored that the data covered the full lower atmosphere over a region containing many large cities, including Shanghai, Tianjin, and Jinan and that they concerned only NO₂ and not other air pollutant forms believed to play a larger role in total population health risk.

8. Our program has conducted some research on indoor air pollution in earlier phases, both in rural and urban settings. These included a multidisciplinary social survey on the human dimensions of air quality and environmental policy implementation in rural Anhui (Alford et al. 2001), coupled with support for epidemiological research on indoor air pollution in the same area (Venness et al. 2001).

9. This does not include the valuation of damages from water pollution, which together with the air pollution categories add up to the 7.7% of GDP noted in the report (World Bank 1997).

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