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COST OF POLLUTION IN CHINA

THE WORLD BANK

污染的负担在中国

CONFERENCE EDITION

COST OF POLLUTION IN CHINA

ECONOMIC ESTIMATES OF PHYSICAL DAMAGES

THE WORLD BANK

THE GOVERNMENT OF THE PEOPLE'S REPUBLIC OF CHINA

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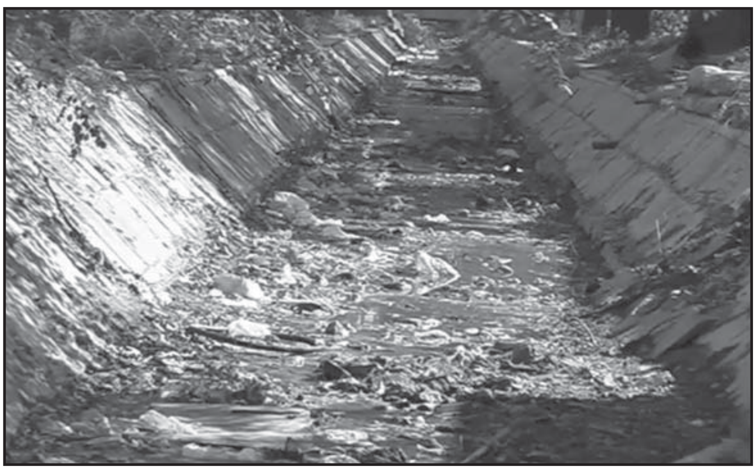
ECONOMIC ESTIMATES OF PHYSICAL DAMAGES



The World Bank
State Environmental Protection Administration, P. R. China

This publication is available online at www.worldbank.org/eapenvironment.

Front cover photos: John D. Liu. From the film "*A Green Call*" prepared by the Environmental Education Media Project in Beijing in cooperation with the World Bank.
Cover design: Circle Graphics, Jostein Nygard

Rural Development, Natural Resources and Environment Management Unit
East Asia and Pacific Region
The World Bank
Washington, D.C.
February, 2007

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Acknowledgments

This Report is the result of a collaborative research effort by a joint Chinese and international expert team being contracted by the World Bank.

In Beijing, the overall team was lead by Guo Xiaomin, a senior advisor to the State Environmental Protection Administration (SEPA). His team included Yu Fang from the China Academy of Environmental Planning (CAEP), who has handled the overall technical coordination, Zhou Guomei coordinated the Valuation of Environmental Health Risk (VEHR) study together with Zhang Kai, Zhou Jun and Wu Yuping from the Policy Research Center for Environment & Economy. Pan Xiaochuan at the Medical College of Peking University lead a team on dose response function development, which included Wang Lihua, and Jiang Jinhua. Monitoring data was provided by the China National Monitoring Centre by Zhuo Jianping, Ling Lixin, Fu Deqing and WuHuaimin. Zhao Yaoming has participated from the Ministry of Agriculture.

A team from the Water Resources and Hydropower Planning and Design Institute of the Ministry of Water Resources (MWR) lead by Li Yuanyuan, which also included Zhou Zhiwei, Cao Jianting and Zhangwei provided assistance on water scarcity subjects. Gao Jun and Xu Ling from the Ministry of Health (MoH) worked on statistical health data. A team from the Rural Water Supply Central Disease Control (CDC) lead by Fan Fucheng and Tao Yong provided assistance on particularly drinking water and partly health related data.

In Shanghai, a team lead by Prof. Peng Xizhe at the Fudan University, included Chen Yan, Tian Wenhua and Cheng Yuan. In Chongqing, a team lead by Ass. Prof. Chen Gangcai at the Chongqing Academy of Environmental Science included Wang Fei, Ran Tao, Zhou Zhien, Liu Lanyu, and Chen Derong in addition to Yang Xioalin, Xiang Xinzhi and Qin Lei from Chongqing CDC and Tang Guil from Chongqing MoH.

International experts have included Haakon Vennemo and Henrik Lindhjem (ECON), Kristin Aunan and Hans Martin Seip (CICERO), Alan Krupnick, Sandy Hoffmann and Michael McWilliams (RFF), Bjorn Larsen and Ramon Ortiz (independent consultants).

At the World Bank, the project was coordinated by Jostein Nygard, task team leader (EASRE) under the overall supervision of Magda Lovei (EASOP). Substantive inputs were provided by Maureen Cropper (DEC), Tamer Samah Rabie (ECSHD), while technical support was provided by Marija Kuzmanovic and Andrew Murray (EASEN/EASRE).

The current report has mainly be written by Maureen Cropper, Tamer Rabie, Haakon Vennemo, Kristin Aunan, Hans Martin Seip, Yu Fang, Guo Xiaoming and Jostein Nygard, while the extensive Chinese expert team has mainly been writing the progress and background reports that this report builds upon. The RFF, Shanghai and Chongqing teams have been writing the "*Willingness to Pay for Reduced Mortality Risk Reduction in Shanghai and Chongqing*" study, which is also being published as a separate World Bank discussion paper report. Mainly based upon work by Bjorn Larsen, a separate discussion paper report "*China Health Effects of Indoor Air Pollution*" is also being published.

Peer reviewers included Chris Nielsen (Harvard University), Hao Jiming (Tsinghua University), Kseniya Lvovsky (World Bank, SASES), Rita Klees (World Bank, ENV), and Anil Markandy (ECSSD). Additional reviews and comments were provided by David Dollar, Bert Hofman and Andres Liebenthal (World Bank, Beijing), Maria Teresa Serra (EASES/EAPVP) Julien Labonne and Jian Xie (EASES/EASRE), Anjali Acharya and Giovanni Ruta (ENV) and Charles E. Di Leva (LEGEN).

Coordination of the study within SEPA, has been made by their Foreign Economic Cooperation Office (FECO) with Wang Xin and Xie Yongming. Personnel within SEPA's Planning and Finance, Pollution Control and Science and Technology departments in addition MoH personnel have reviewed the report extensively.

The report was edited by Robert Livernash, consultant. Circle Graphics designed and managed desktopping. Production was supervised by Jaime Alvarez. Photos provided by John Liu, the Environmental Education Media Project, from a World Bank-contracted film "*A Green Call*". Chinese translation was provided by the translation desk at SEPAs Department of International Cooperation.

Finally, we would like to express our gratitude to the Government of Norway and Finland, which provided the main trust funds (TFESSD) to carry out the study. The study was also supported by the World Bank's own funding.

Abbreviations and Acronyms

ACS	American Cancer Society
AHC	Adjusted Human Capital
BOD	Biological Oxygen Demand
BOH	Bureau of Health (at local levels)
CAEP	Chinese Academy for Environmental Planning
CAES	Chongqing Academy of Environmental Sciences
CDC	Center for Disease Control and Prevention
CECM	Chinese Environmental Cost Model
CEVD	Cerebrovascular Disease
CNHS	China National Health Survey
CO	Carbon Monoxide
COD	Chemical Oxygen Demand
COI	Cost of Illness
COPD	Chronic Obstructive Pulmonary Disease
CSMI	Clear Water and Sewage Mixed Irrigation
CV	Contingent Valuation
CVD	Cardiovascular Disease
DALY	Disability-Adjusted Life Year
DC	Dichotomous Choice Method
DSP	Disease Surveillance Point
ECM	Environmental Cost Model
EU	European Union
EV	Emergency Visit
GDP	Gross Domestic Product
GIOV	Gross Industrial Output Value
HEI	Health Effects Institute
HH	Household
ICD	International Classification of Disease

IWQI	Integrated Water Quality Index
MoA	Ministry of Agriculture
MoH	Ministry of Health
MWR	Ministry of Water Resources
NAPAP	National Acid Precipitation Assessment Program
NBS	National Bureau of Statistics
NOx	Nitrogen Oxides
O_3	Ozone
OPV	Outpatient Visit
OR	Odds Ratio
PC	Payment Card Method
PM	Particulate Matter
PM10	Particulate Matter of Less than 10 μm in diameter
PPP	Purchasing Power Parity
PSI	Pure Sewage Irrigation
QALY	Quality Adjusted Life Year
RD	Respiratory Disease
RFF	Resources for the Future
RMB	Chinese Currency, Yuan
RR	Relative Risk
SCE	Standard Coal Equivalent
SEPA	State Environmental Protection Administration
SO_2	Sulphur Dioxide
TSP	Total Suspended Particulates
TVEs	Town and Village Enterprises
UNEP	United Nations Environmental Programme
USEPA	United States Environmental Protection Agency
VEHR	Valuation of Environmental Health Risk
VSL	Value of Statistical Life
WHO	World Health Organization
WTP	Willingness to Pay

Foreword to the Conference Edition

This is a draft edition of the *Cost of Pollution in China: Economic Estimates of Physical Damages* report, which will be presented at the international conference on Sustainable Development in Beijing, China on March 2, 2007. The purpose of this *conference edition* is to present the findings of the studies undertaken in China over the past about 3 years as well as to obtain relevant comments and feedback from the conference participants that could be included in the final edition of the report.

This report traces its origin to 1997, when the World Bank published the *China 2020 – Clear Water Blue Skies* report. This work underscored the economic implications of environmental degradation by estimating that the cost of air and water pollution in China is between 3.5 and 8 percent of GDP. Following these findings, the Chinese government requested the World Bank to collaborate with a number of Chinese and international research institutes to develop an environmental cost model (ECM) using methodologies specific to the China context.

This work includes an in-depth review of international ECM studies, and development and application of new methodologies (and software) for annual estimations of water and air pollution in China at both central and local levels. The aim of this work is to increase awareness of the economic impacts of air and water pollution in China, to provide relevant policy information to decision makers and to enable the Chinese government to make optimal resource allocations for environmental protection.

Prior to the publication of this report, comprehensive comments have been received by both the Chinese Government, particularly the State Environmental Protection Administration (SEPA) and independent Chinese and Non-Chinese reviewers. Some of the subjects that have been carefully developed during the course of implementation, including certain physical impact estimations as well as economic cost calculations at local levels have been left out of this conference edition due to still some uncertainties about calculation methods and its application. How to possibly make use of these materials will be continuously worked on during and after the conference. Moreover, the comprehensive reference material that has been developed by joint Chinese and International expert team (including progress reports and various background reports), is going to be attached in a CD-ROM in the final edition.

Wish you good reading of this edition and looking forward to receiving your comments.

Report Authors
February 2007

Executive Summary

In recent decades, China has achieved rapid economic growth, industrialization, and urbanization. Annual increases in GDP of 8 to 9 percent have lifted some 400 million people out of dire poverty. Between 1979 and 2005, China moved up from a rank of 108th to 72nd on the World Development Index. With further economic growth, most of the remaining 200 million people living below one dollar per day may soon escape from poverty. Although technological change, urbanization, and China's high savings rate suggest that continued rapid growth is feasible, the resources that such growth demands and the environmental pressures it brings have raised grave concerns about the long-term sustainability and hidden costs of growth. Many of these concerns are associated with the impacts of air and water pollution.

Rapid Economic Growth Has Had Positive Environmental Impacts but Also Created New Environmental Challenges

Considering China's strong economic growth over the last 20–25 years, there is no doubt that it has had positive impacts on the environment. Alongside economic growth, *technology improvements* over this period have created much-improved resource utilization. Energy efficiency has improved drastically—almost three times better utilization of energy resources in 2000–02 compared to 1978. As a result of the *changing industrial structure*, the application of cleaner and more energy-efficient technologies, and pollution control efforts, ambient concentrations of particulate matter (PM) and sulfur dioxide (SO_2) in cities have gradually decreased over the last 25 years. Implementation of *environmental pollution control policies*—particularly command-and-control measures, but also economic and voluntarily measures—have contributed substantially to leveling off or even reducing pollution loads, particularly in certain targeted industrial sectors.

At the same time, new environmental challenges have been created. Following a period of stagnation in energy use during the late 1990s, total energy consumption in China has increased 70 percent between 2000 and 2005, with coal consumption increasing by 75 percent, indicating an increasingly energy-intensive economy over the last few years. Moreover, between 2000 and 2005, air pollution emissions have remained constant or, in some instances, have increased. The assessment at the end of the tenth five-year plan (2001–05) recently concluded that China's emissions of SO_2 and soot were respectively 42 percent and 11 percent higher than the target set at the beginning of the plan. China is now the largest source of SO_2 emissions in the world. Recent trends in energy consumption, particularly increased coal use, provide a possible explanation for the increase in SO_2 emissions.

Water pollution is also a cause for serious concern. In the period between 2001 and 2005, on average about 54 percent of the seven main rivers in China contained water deemed unsafe for human consumption. This represents a nearly 12 percent increase since the early 1990s. The most polluted rivers occurred in the northeast in areas of high population density. The trends in surface water quality from 2000 to 2005 suggest that quality is worsening in the main river systems in the North, while improving slightly in the South. This may partly be the result of rapid urbanization (the urban population increased by103 million countrywide from 2000 to 2005), which caused COD loads from urban residents to increase substantially and, hence, surpass the planned targets for 2005. Rapid industrialization probably also plays a part.

Northern China Bears a Double Burden from Air and Water Pollution

While the most populous parts of China also have the highest number of people exposed to air pollution, it is striking that the areas with the highest per capita exposure are almost all located in northern China (Qinghai, Ningxia, Beijing, Tianjin, Shaanxi, and Shanxi). The exception is Hunan, which is located in the South. In Figure 1, the color of the provinces on the map shows the percentage of the urban population exposed to air pollution, while the bars indicate the absolute number of people exposed.

Similarly, the most severely polluted water basins—of the Liao, Hai, Huai, and Songhua rivers—are also located in northern China (see figure 2 for surface water quality). North China also has serious water scarcity problems. Some provinces—including Beijing, Shanxi, Ningxia, Tianjin, and Jiangsu—seem to face the double burden of exposure to high levels of both air and water pollution. However, while air pollution levels may be directly associated with population

FIGURE 1. Urban Population Exposed to PM10 levels, 2003

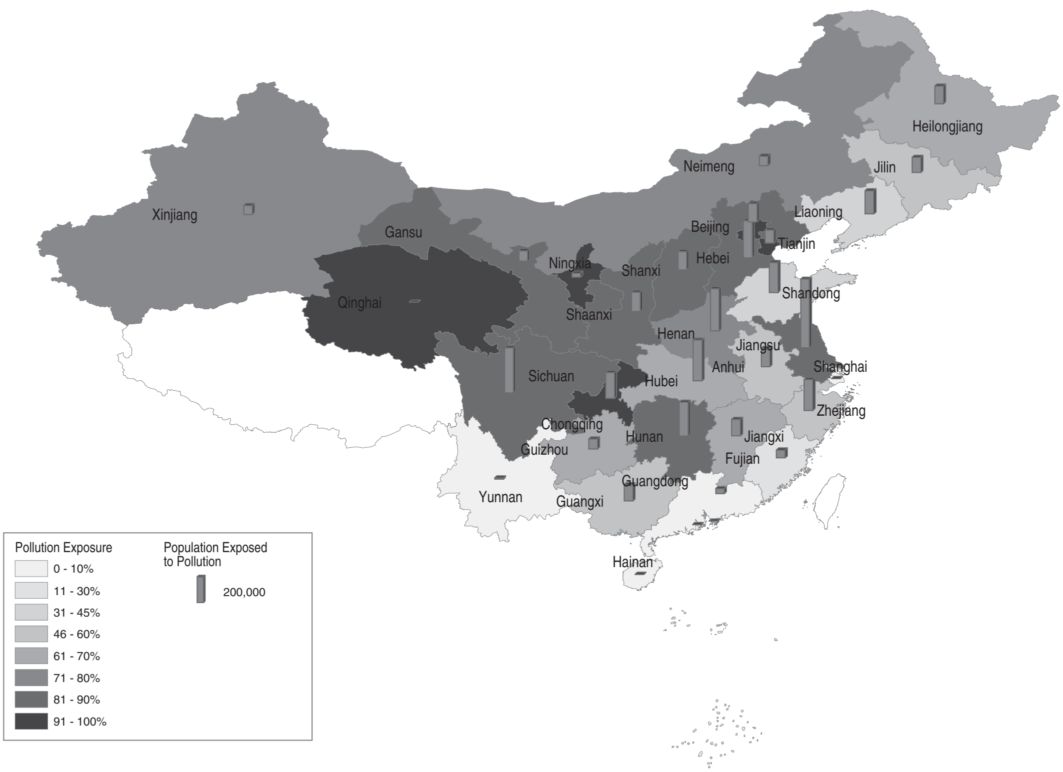
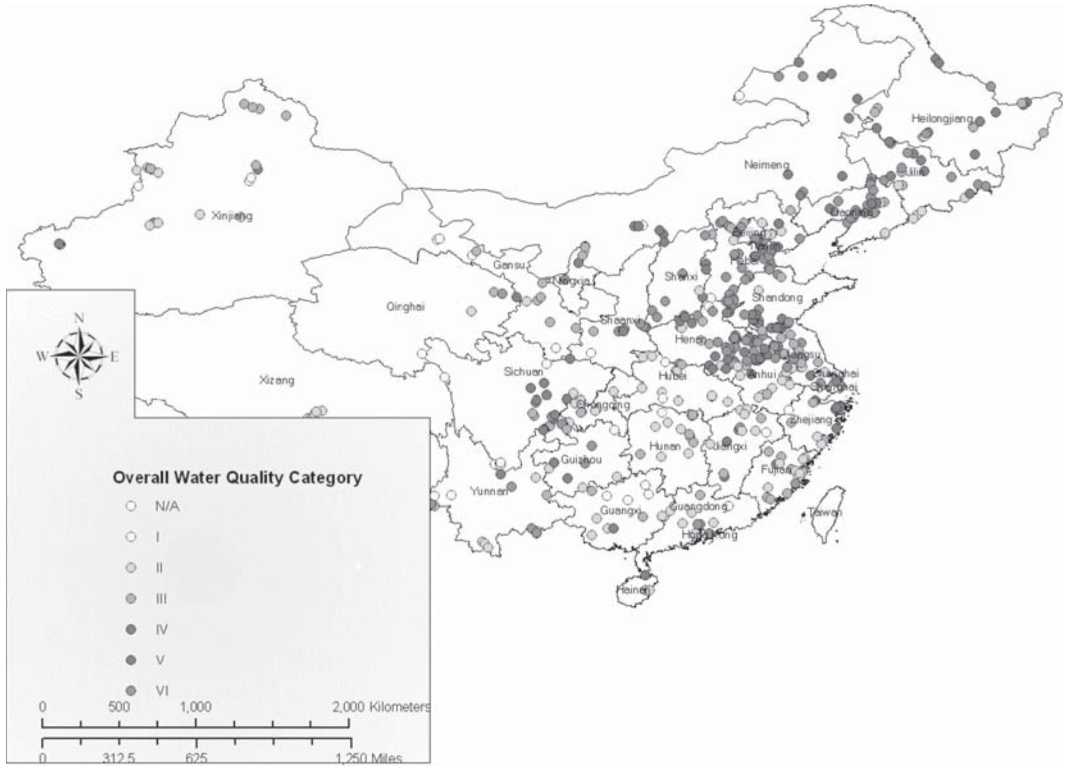


FIGURE 2. Water Quality Levels, 2004



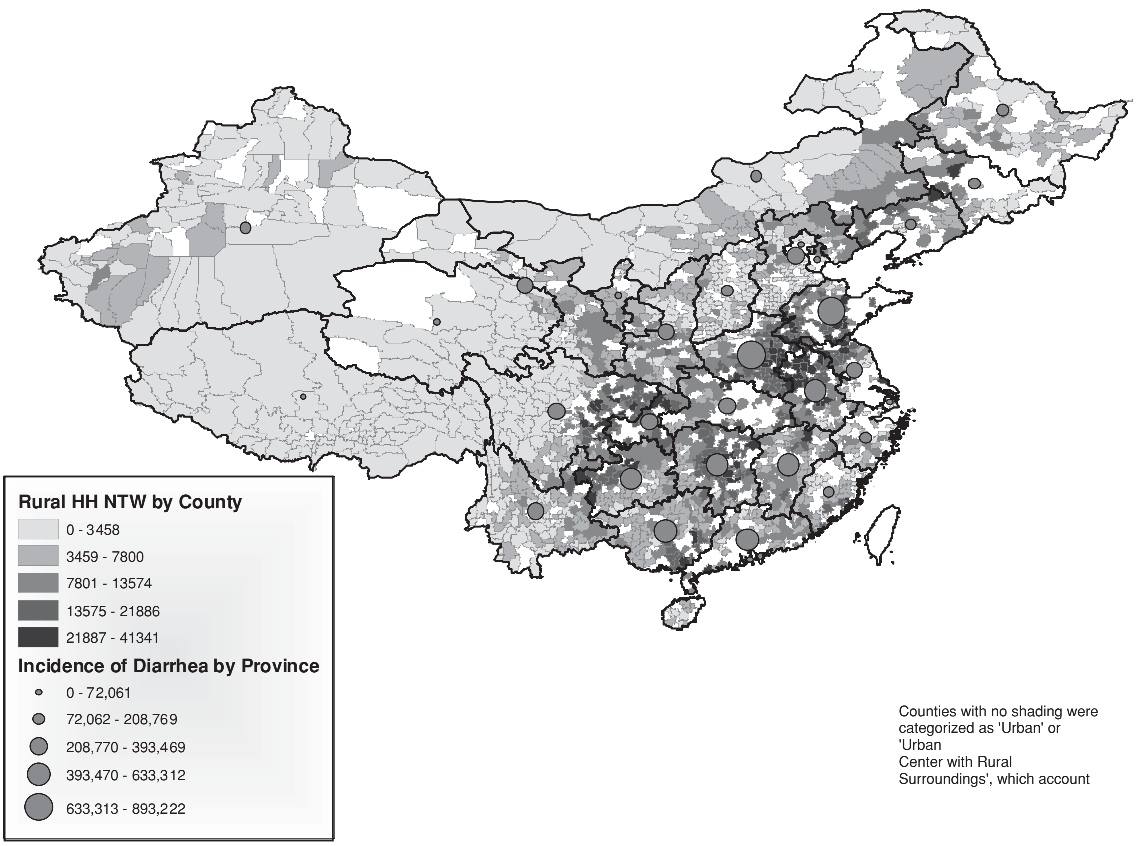
exposure, the same does not necessarily apply to surface water pollution. This is because populations generally have different drinking water sources that may allow them to escape high levels of contamination. About 115 million people in rural China rely primarily on surface water as their main source of drinking water. Surface water as a drinking water source is more vulnerable to possible pollution compared to other, safer drinking sources.

Air and Water Pollution have Severe Health Impacts

According to conservative estimates, the economic burden of premature mortality and morbidity associated with air pollution was 157.3 billion yuan in 2003, or 1.16 percent of GDP. This assumes that premature deaths are valued using the present value of per capita GDP over the remainder of the individual's lifetime. If a premature death is valued using a value of a statistical life of 1 million yuan, reflecting people's willingness to pay to avoid mortality risks, the damages associated with air pollution are 3.8 percent of GDP. These findings differ in two important ways from previous studies of the burden of outdoor air pollution in China. First, they are based on Chinese exposure-response functions, as well as on the international literature; and second, they are computed for individual cities and provinces. Previous estimates by WHO (Cohen et al. 2004) were based on the assumption that increases in PM beyond 100 $\mu g/m^3$ of PM_{10} caused no additional health damage.(In the base case considered by WHO, relative risk does not increase beyond 50 μg/m3 of $PM_{2.5}$, which is approximately equivalent to 100 μg/m3 of PM_{10}.) This assumption implies that the WHO estimates cannot be used to evaluate the benefits of specific urban air pollution control policies.

Two-thirds of the rural population is without piped water, which contributes to diarrheal disease and cancers of the digestive system. The cost of these health impacts, if valued using a VSL of 1 million, are 1.9 percent of rural GDP. Analysis of data from the 2003 *National Health Survey* indicates that two-thirds of the rural population does not have access to piped water. The relationship between access to piped water and the incidence of diarrheal disease in children under the age of 5 confirms this finding: the lack of access to piped water is significantly associated with excess cases of diarrheal disease and deaths due to diarrheal disease in children under 5 years of age. Although there are many indications that surface and drinking water pollution problems contribute to serious health impacts, the lack of monitoring data on specific pollutants and data on household behavior regarding avoiding exposure to polluted drinking water make it difficult to quantify all of the health effects of water pollution. Specifically, the lack of exposure data makes quantifying the relationship between chemical and inorganic pollution and the incidence of chronic diseases almost impossible. Preliminary estimates suggest that about 11 percent of cases of cancer of the digestive system may be attributable to polluted drinking water. More

FIGURE 3. Rural Households with No Access to Piped Water & Diarrhea Incidence



attention, however, needs to be given at the policy level to reinforcing the surveillance capacity for chronic exposures and disease incidence.

Health is Highly Valued by the People in China

The mortality valuation surveys conducted in Shanghai and Chongqing as part of this study suggest that people in China value improvements in health beyond productivity gains. The value of a statistical life estimated in these surveys—the sum of people's willingness to pay for mortality risk reductions that sum to one statistical life—is approximately 1 million yuan. This number supports results of other studies, which suggest that the value of an avoided death is greater than what is implied by the adjusted human capital approach, which is approximately 280,000 Yuan in urban areas. Evaluation of the health losses due to ambient air pollution using willingness-to-pay measures raises the cost to 3.8 percent of GDP.

It is remarkable that the willingness to pay is about the same in locations as different as Shanghai and Chongqing, which differ greatly in per capita GDP with a ratio as high as 5:1. (However, sample per capita incomes showed a more modest ratio of 2:1.) Furthermore, these new findings illustrate that the urban Chinese population has a willingness to pay to reduce mortality risk comparable in PPP terms to the levels seen in several developed countries with much higher per capita incomes. This means that the Chinese people highly value their health status and their longevity.

China's Poor Are Disproportionately Affected by Environmental Health Burdens

Although the objective of this study was not to compare the impacts of air and water pollution on the poor versus the non-poor, the findings suggest that environmental pollution falls disproportionately on the less economically advanced parts of China, which have a higher share of poor populations. As shown in Figure 1, Ningxia, Xinjiang, Inner Mongolia, and other low-income provinces are more affected by air pollution on a per capita basis than high-income provinces such as Guangdong and other provinces in the southeast.

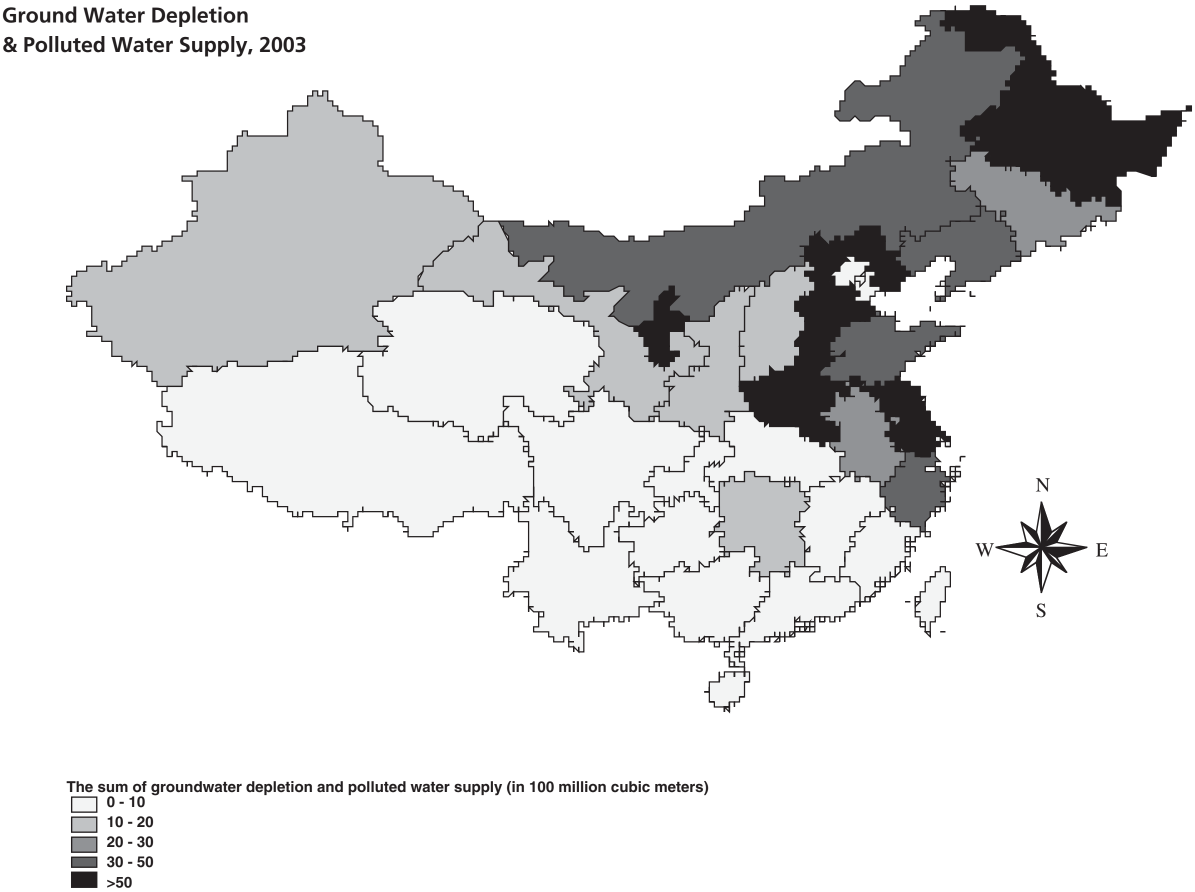
From another perspective, analysis of the 2003 *National Health Survey* showed that 75 percent of low-income households in rural China with children under 5 years of age have no access to piped water, compared to 47 percent in the higher-income categories. This implies that low-income households rely more on other drinking water sources. In fact, about 32 percent of households within the lowest income quartile rely primarily on surface water as their primary source of drinking water, compared to 11 percent in the highest income quintile. This means that the rural poor are at a substantially higher risk from surface water pollution than the non-poor.

The fact that water quality in the North is worse than in the South may explain the slightly higher diarrheal prevalence seen in lower income groups in northern China (2.1 percent) compared to southern China (1.9 percent). However, when focusing on differences between income groups in the North, the data clearly show that the poor (lowest income quartile) have a much higher diarrheal prevalence (2.4 percent) in households using surface water compared to the highest income groups, where no diarrhea cases have been recorded.

Pollution Exacerbates Water Scarcity, Costing 147 Billion Yuan a Year

Water scarcity is a chronic problem, especially in the North. It is closely related to problems of water pollution. Surface water pollution has put pressure on the use of groundwater for agricultural and industrial purposes. The depletion of

FIGURE 4. Groundwater Depletion and Polluted Water Supply



nonrechargeable groundwater in deep freshwater aquifers imposes an environmental cost, since it depletes a nonrenewable resource and increases future costs of pumping groundwater. It can also lead to seawater intrusion and land subsidence.

Estimates of the cost of groundwater depletion suggest that it is on the order of 50 billion yuan per year, while estimates of the costs of using polluted water to industry are comparable in magnitude, bringing the overall cost of water scarcity associated with water pollution to 147 billion yuan, or about 1 percent of GDP. These new findings indicate that the effects of water pollution on water scarcity are much more severe than previous studies have estimated.

Air and Water Pollution Cause Significant Crop and Material Damage

This study makes clear that the impacts of air and water pollution on health are severe in both absolute and in economic value terms. Although we acknowledge that not all non-health-related impacts can be quantified, the impacts of pollution on natural resources (agriculture, fish and forests) and manmade structures (e.g. buildings) are estimated to account for substantially lower damages in economic terms.

Acid Rain costs 30 billion yuan in crop damage and 7 billion in material damage annually. It is estimated that acid rain, caused mainly by increased SO_2 emissions due to increased fossil fuel use—causes over 30 billion yuan in damages to crops, primarily vegetable crops (about 80 percent of the losses). This amounts to 1.8 percent of the value of agricultural output. Damage to building materials in the South imposed a cost of 7 billion yuan on the Chinese economy in 2003. In addition to the human health effects reported above, these damages provide an additional impetus for controlling SO_2. Damages to forests could not be quantified due to lack of monitoring data in remote areas and adequate dose-response functions.

Six provinces account for 50 percent of acid rain effects. The burden of damages from acid rain is also unevenly distributed. Over half of the estimated damages to buildings occur in three provinces: Guangdong (24 percent), Zhejiang (16 percent), and Jiangsu (16 percent). Almost half of the acid rain damage to crops occurs in three provinces: Hebei (21 percent), Hunan (12 percent), and Shandong (11 percent). However, the impacts of acid rain extend across international boundaries and also affect neighboring countries.

Irrigation with polluted water costs 7 billion yuan per year. This study has quantified part of the damage caused by the use of polluted water for irrigation in agriculture and a portion of the impact of water pollution on fisheries. The impact of irrigating with polluted water in designated wastewater irrigation zones—considering only the impact on yields and produce quality, but not on human health—was estimated to reach 7 billion yuan in 2003.

The cost to fisheries is estimated at 4 billion yuan. The impact of acute water pollution incidents on commercial fisheries is estimated at approximately 4 billion yuan for 2003. The impact of chronic water pollution on fisheries could not be estimated for lack of exposure data as well as adequate dose-response information.

Air Pollution Poses a Large Health Risk in Urban Areas and Water Pollution a Significant Health Risk in Rural Areas

The figures presented in the summary table at the end of this chapter suggest that outdoor air pollution poses a very serious problem in urban areas. This is not surprising when one compares the levels of ambient PM_{10} in Chinese cities with other large cities across the world. With annual average PM_{10} concentrations of over 100Ìg/m^3, several selected cities in both northern and southern China are among the most polluted cities in the world (see figure 5).

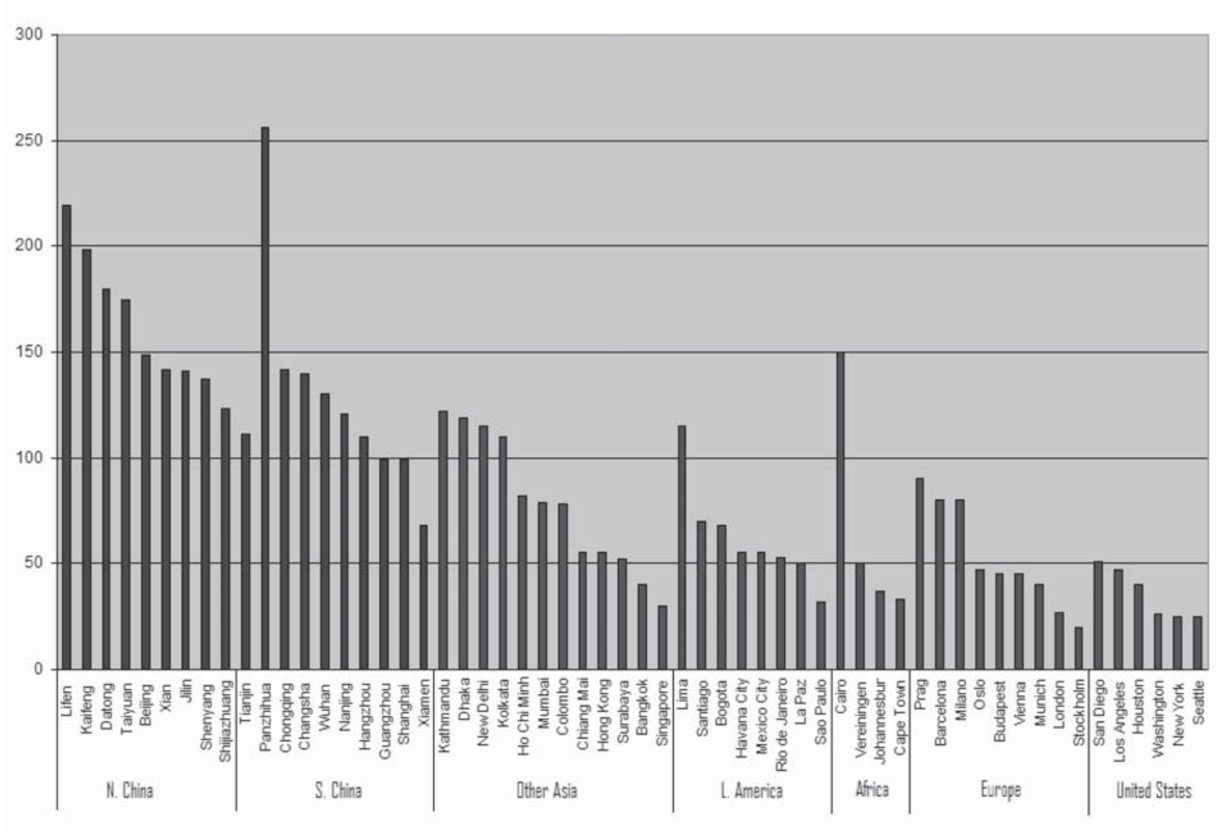
Although the health damages associated with water pollution are smaller, in total, and as a percent of rural GDP, they are still 0.3 percent of *rural* GDP if conservatively valued and 1.9 percent of rural GDP when valued using a 1 million yuan VSL. Both figures ignore the morbidity associated with cancer and therefore underestimate the health costs associated with water pollution. However, relative to other developing countries, China's diarrheal prevalence in rural areas is quite low, actually lower than in countries where a larger percentage of the rural population has access to piped water supply (see figure 6).

The Benefits of Sound Policy Interventions May Exceed the Costs

This study report shows that the total cost of air and water pollution in China in 2003 was 362 billion yuan, or about 2.68 percent of GDP for the same year. However, it should be noted that this figure reflects the use of the adjusted human capital approach, which is widely used in Chinese literature, to value health damages. If the adjusted human capital approach is replaced by the value of a statistical life (VSL) based on studies conducted in Shanghai and Chongqing, the amount goes up to about 781 billion yuan, or about 5.78 percent of GDP.

Setting priorities for cost-effective interventions. Interventions to improve the environment in China are likely to yield positive net benefits. Indeed, one of the advantages of the environ-

FIGURE 5. Annual average PM10 concentrations observed in selected cities worldwide, 2004, 2005

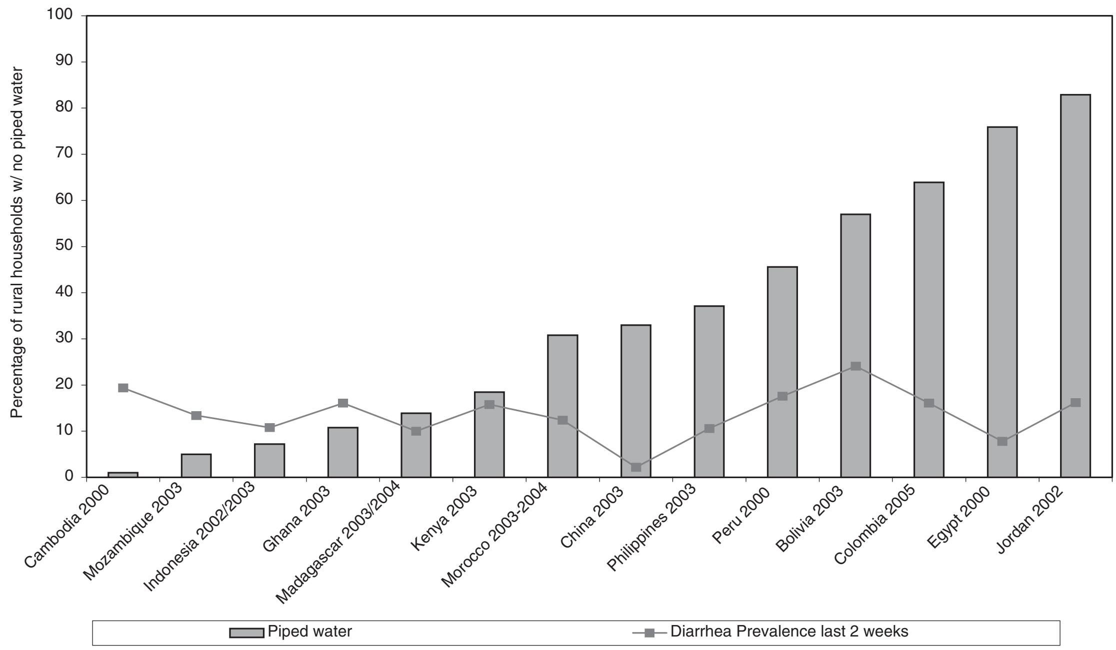


Source: China Environmental Yearbook 2005 and WHO 2005.

mental cost model developed in this project is that it can be used to evaluate the benefits of specific pollution-control policies and assist in designing and selecting appropriate targeted intervention policies. Once the impact on ambient air quality of a policy to reduce particulate emissions has been calculated, the tools used to calculate the health damages associated with particulate emissions can be used to compute the benefits of reducing them. To illustrate, researchers have examined the costs and impacts on ambient air quality of measures to control SO_2 emissions and fine particles in Shijiazhuang, the capital of Hebei Province (Guttikunda et al. 2003). The monetized value of the health benefits associated with each measure could be calculated, using the techniques developed in this study, and compared with the costs.

Targeting high-risk areas. The findings from this project suggest that a focus on northern China is essential, particularly the North China Plain and areas located northeast and northwest of the plain, where the study shows that there is a double burden from both air and water pollution. This problem is further magnified by the presence of disparities between the poor and non-poor. On this basis, it seems relevant that stronger policy interventions should be developed to address air and water pollution problems. In addition, these efforts should be complemented with emphasis on improving

Figure 6. Diarrheal Prevalence and Access to Piped Water Supply



Source: ORC Macro, 2006. MEASURE DHS STATcompiler. http://www.measuredhs.com, July 3 2006.

access to clean water, with a specific focus on the lowest income groups.

Responding to people's concerns. This study suggests that the Chinese value the avoidance of health risks beyond productivity gains. This implies that people's preference for a clean environment and reduced health risks associated with pollution are stronger than past policies appear to have acknowledged. Growing concerns about the impacts of pollution are increasingly expected to guide national policies as well as local actions. Public disclosure of environmental information such as emissions by polluting enterprises, as well as ambient environmental quality data by local authorities, could be an important tool for responding to people's concerns and creating incentives for improving local conditions.

Addressing the information gap. Past policies and decisions have been made in the absence of concrete knowledge of the environmental impacts and costs. By providing new, quantitative information based on Chinese research under Chinese conditions, this study has aimed to reduce this information gap. At the same time, it has pointed out that substantially more information is needed in order to understand the health and non-health consequences of pollution, particularly in the water sector. It is critically important that existing water, health, and environmental data be made publicly available so the fullest use can be made of them. This would facilitate conducting studies on the impacts of water pollution on human and animal health. Furthermore, surveillance capacity at the local and national levels needs to be expanded to improve the collection of environmental data, especially data on drinking water quality. These efforts will further improve the analysis begun in this project.

Developing an environmental-health action plan. At present, an environmental-health action plan is being jointly drafted by the State Environmental Protection Administration (SEPA) and the Ministry of Health (MoH). This plan should take into consideration the mortality and morbidity impacts from water and air pollution presented in this report. The plan should include a focus on the geographical areas identified in northern China, where there is a double burden of both air and water pollution. Furthermore, particular focus should be put on areas where poor populations are adversely affected from lack of access to clean water and sanitation.

1

Overview

AIR AND WATER POLLUTION IN CHINA

In the last 25 years, China has achieved rapid economic growth, industrialization, and urbanization, with annual increases in GDP of 8 to 9 percent. During the same period, advances in technology and economic efficiency, coupled with pollution control policies, have positively affected air and water pollution loads. However, great challenges remain in further improving China's environmental status.

To illustrate, China has not been able to meet 10 of its 13 critical 10th five-year-plan targets for air and water pollution control (see table 1.1). The most pressing off-target performance is the drastic increase in industrial-based SO_2 emissions, which has reversed the downward trend in SO_2 levels, and degraded air quality and the increase in domestic COD loads, which have caused water quality to deteriorate.

China is the world's second largest energy consumer after the United States. Almost 68 percent of its energy comes from coal, much of which is

TABLE 1.1 Environmental Targets for the 10th Five Year Plan vs. Environmental Performance (million tons)

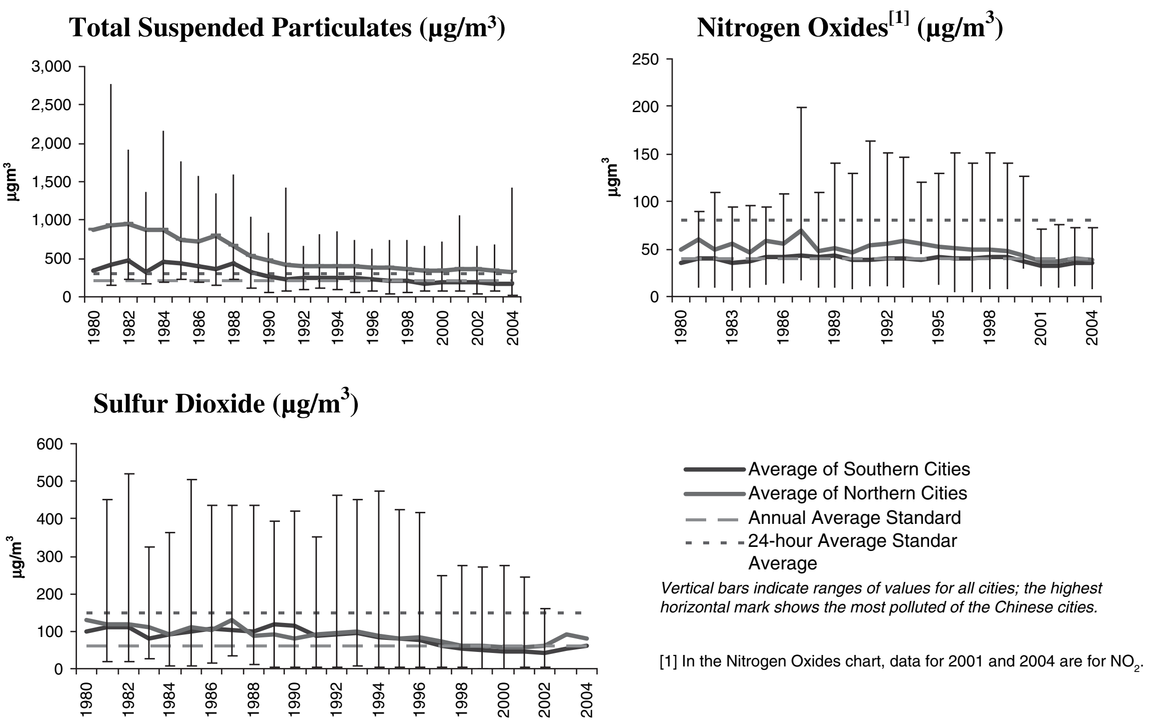
Indicators	Actual 2000	Planned 2005	Actual 2005 (completed by 6/17/06)	Comparison with Planned 2005 (+/– %)
Air Pollution				
SO_2 emissions	19.9	17.9	25.5	42
Industry	16.1	14.5	21.7	50
Domestic	3.8	3.5	3.8	9
Soot Emissions	11.7	10.6	11.8	11
Industry	9.5	8.5	9.5	12
Domestic	2.1	2.1	2.3	10
Industrial Dust Emissions	10.9	8.98	9.1	1
Water Pollution				
COD discharge	14.5	13.0	14.1	8
Industry	7.0	6.7	5.5	–18
Domestic	7.4	6.5	8.6	32
Ammonia Nitrogen	1.8	1.65	1.5	–9
Industry	0.8	0.7	0.525	–25
Domestic	1.1	0.9	0.973	8

Source: Estimations based upon China Environmental Yearbook 2001 and 2006, the 10th Five Year Plan for Environmental Protection and status of the China environment report, 2005 burned in thermal power plants or in industrial boilers. This has led to continuously high levels of SO_2 and particulate air pollution. In addition, water pollution and water scarcity problems are also very severe, particularly in North China, where the region faces some of the most severe water quality and quantity challenges in the world today. This section provides a brief overview of these challenges.

Air Pollution Trends

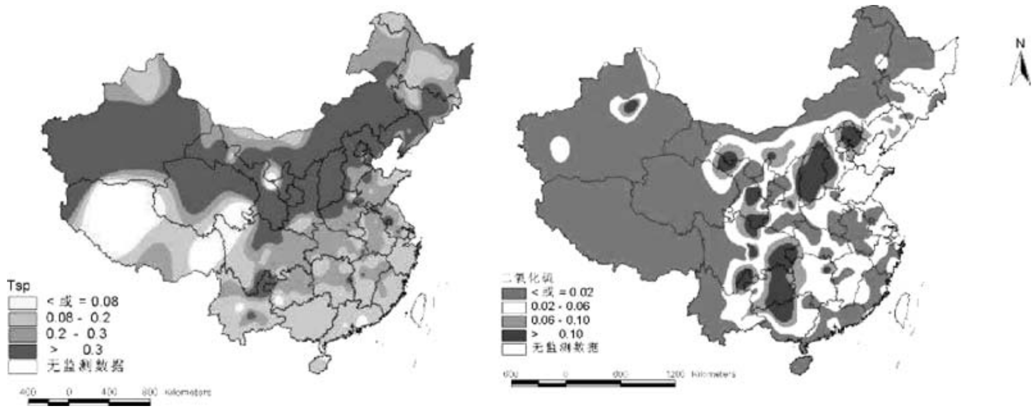
Although levels of SO_2 and particulates have declined since the 1980s, China's cities still rank among the most polluted in the world. Figure 1.1 shows trends in annual average total suspended particulates (TSP, SO_2, and NOx in large and medium-sized Chinese cities, beginning in 1980). (The averages in each year are arithmetic averages—unweighted by population—of available readings for "major cities." The set of cities varies from 53 to 97, depending on the year.) Separate averages are reported for northern and southern cities. Suspended particulate levels are higher in northern cities, due in part to industrial activity, but also to geographic and meteorological conditions that make these cities more vulnerable to particulate pollution than cities in the south of China, holding emissions constant (Pandey et al. 2005). In both northern and southern cities, particulate concentrations show a downward trend from 1980 until the early 1990s and then remain relatively flat. Sulfur dioxide and NOx concentrations also show a downward trend

FIGURE 1.1 Ambient Air Pollution Levels in China's Major Cities (annual averages) Compared to Chinese Class II Air Quality Standards



Source: China Environmental Year Books 2004 & 2005

FIGURE 1.2 TSP and SO_2 Concentrations in China, 2002



Source: Abstracted from www.sepa.gov.cn/

in northern cities. Since 2003, however, NOx and particularly SO_2 concentrations have increased.

When measured in terms of the number of cities violating Chinese air quality standards, air quality has shown some improvement since 1999. Table 1.2 shows the number of cities violating at least one air quality standard (cities classified as Grade III or worse than Grade III) since 1999. The number of cities worse than Grade III has declined steadily since 1999. Nevertheless, in 2005 about 50 percent of China's cities still did not meet air quality standards.

Table 1.3 presents the distribution of monitored cities by PM_{10} and SO_2 levels in 2003 and 2004. In 2003, 53 percent of the 341 monitored cities—accounting for 58 percent of the country's urban population—reported annual average PM_{10} levels in excess of 100 μg/m^3, which is twice the U.S. annual average standard. Twenty-one percent of cities reported annual average levels in excess of 150 μg/m^3. Only 1 percent of the country's urban population lives in cities with annual average PM_{10} levels below 40 μg/m^3.

Sulfur dioxide levels in cities measure up better in terms of international standards. In 2003, almost three-quarters of cities had sulfur dioxide levels below the U.S. annual average standard (60 μg/m^3), suggesting that particulate air pollution is likely to be a more important health concern in the future.

A direct consequence of air pollution from SO_2 and NO_X is acid rain, which remains a serious

TABLE 1.2 Trends in Air Quality in China's Cities (%)

Air Quality Standards	1999	2000	2001	2002	2003	2004	2005
Grade II (Up to the standard)	33	37	34	36	42	39	52
Grade III	26	30	33	34	31	41	38
Worse than grade III	41	33	33	28	27	20	10

Source: Status of China Environment reports 1999–2005

TABLE 1.3 Distribution of PM_{10} and SO_2 Levels in 341 Cities, 2003 and 2004

	% of Cities	
Distribution of PM_{10} Levels	**2003**	**2004**
$PM_{10} \leq 100\ \mu g/m^3$	46	47
$100 < PM_{10} \leq 150\ \mu g/m^3$	33	39
$PM_{10} > 150\ \mu g/m^3$	21	14
Distribution of SO_2 Levels		
$SO_2 \leq 60\ \mu g/m^3$	74	74
$60 < SO_2 \leq 100\ \mu g/m^3$	14	17
$SO_2 > 100\ \mu g/m^3$	12	9

Source: China Environmental Yearbooks 2004 and 2005.

problem in China. Figure 1.3 shows the distribution of rainfall by pH level in China in 2001, 2003, and 2005. The problem remains serious in the south and southeastern portions of the country. As illustrated below, there are some indications that the main areas affected are gradually moving from southwest to southeast. Over half of China's sulfur dioxide emissions come from electric utilities (Sinton, 2004). Total sulfur dioxide emissions declined in the late 1990s, largely due to stricter standards on emissions of SO_2 by coal-fired power plants and to the "Two Zones" control program designed to reduce acid rain by controlling SO_2 emissions in cities with high ambient SO_2 levels (see the second map in figure 1.2 and the maps in figure 3). However, recent data (see table 1.1) suggest that sulfur dioxide emissions are increasing due to the high demand for coal in a rapidly growing economy. Emissions in 2005 were over 25 million tons, 28 percent higher than in 2000, and 42 percent higher than the 2005 target.

Despite increased SO_2 emissions over the last three years (up 32 percent from 2001 to 2005), it should be noted that the number of cities reaching acceptable SO_2 concentration standards (i.e. reaching class II) has in fact increased in the SO_2 control zone and remained about the same in the acid rain control zone (see table 1.4). This may indicate that SO_2 emission from high point sources have increased, while emissions from low point sources and area sources have decreased.

Water Pollution Trends and Quality

Surface water quality in China is poor in the most densely populated parts of the country, in spite of increases in urban wastewater treatment capacity. Water quality is monitored by the State Environmental Protection Administration (SEPA) in about 500 river sections and by the Ministry of Water Resources in more than 2,000 sections across the main rivers. It is classified into one of five categories based on concentrations of the 30 substances listed in Annex 2. Recent trends suggest that quality is worsening in the main river systems in the North, while improving in the South (see figure 1.4). For all the five main river systems in the North (Songhua, Liao, Hai, Huai, and Huang rivers), sections with class IV to VI ranked

FIGURE 1.3 Distribution of Acid Rain in China, 2001, 2003, and 2005

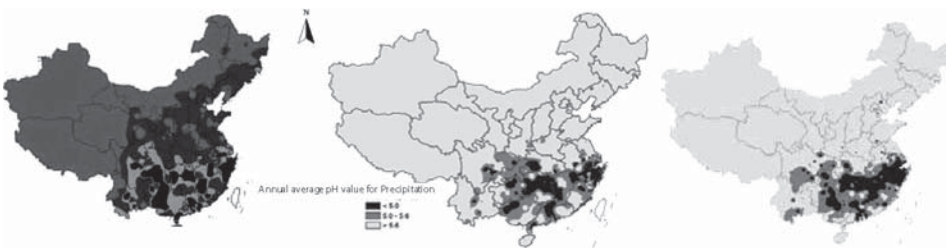
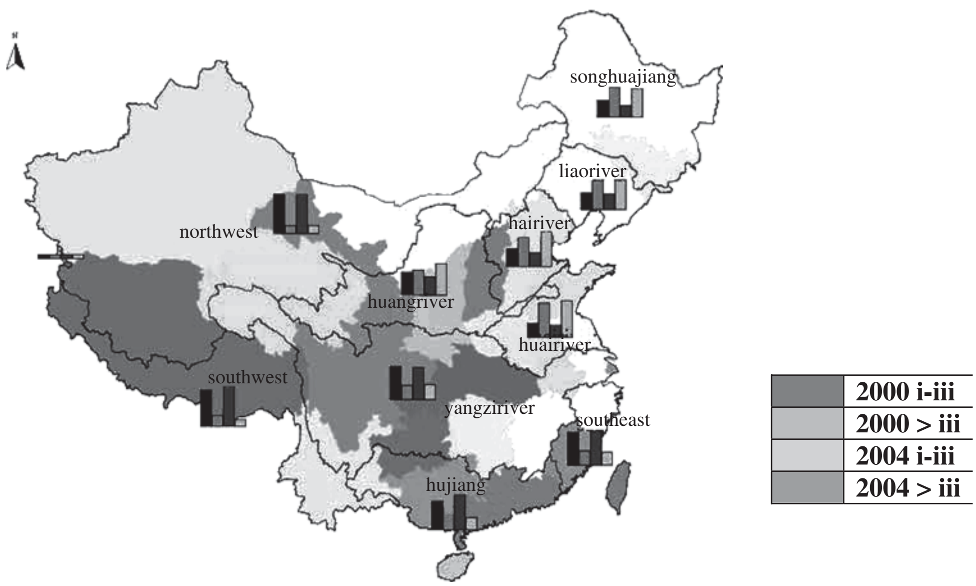


TABLE 1.4 Distribution of SO_2 Levels Among Cities in the Two Air Pollution Control Zones, 1998–2005 (in %)

SO_2 Concentrations	1998	2000	2002	2003	2004	2005
In the SO_2 control zone:						
Reaching Class II standards: ($SO_2 \leq 0.6$ mg/m^3)	33	48	41	39	41	45
Reaching Class III standards: (0.06 mg/m^3 < $SO_2 \leq 0.10$ mg/m^3)	30	25	31	25	30	34
Below Class III standards: (SO_2 > 0.10 mg/m^3)	37	27	28	36	29	21
In the acid rain control zone:						
Reaching Class II standards: ($SO_2 \leq 0.6$ mg/m^3)	70	81	79	75	73	74
Reaching Class III standards: (0.06 mg/m^3 < $SO_2 \leq 0.10$ mg/m^3)	14	6	14	15	20	22
Below Class III standards: (SO_2 > 0.10 mg/m^3)	16	13	7	10	7	4

Source: Status of China Environment reports 2000–05

FIGURE 1.4 Surface Water Quality, 2000 and 2004



Source: China—Water Quality Management—Policy and Institutional Considerations (World Bank, 2006) water—i.e., non-potable water sources, but that may be used by industry (class IV) and agriculture (class V)—increased, while the better class I–III ranked water—i.e. suitable for drinking water, swimming and household use, and which also can support aquatic life—increased in the South.

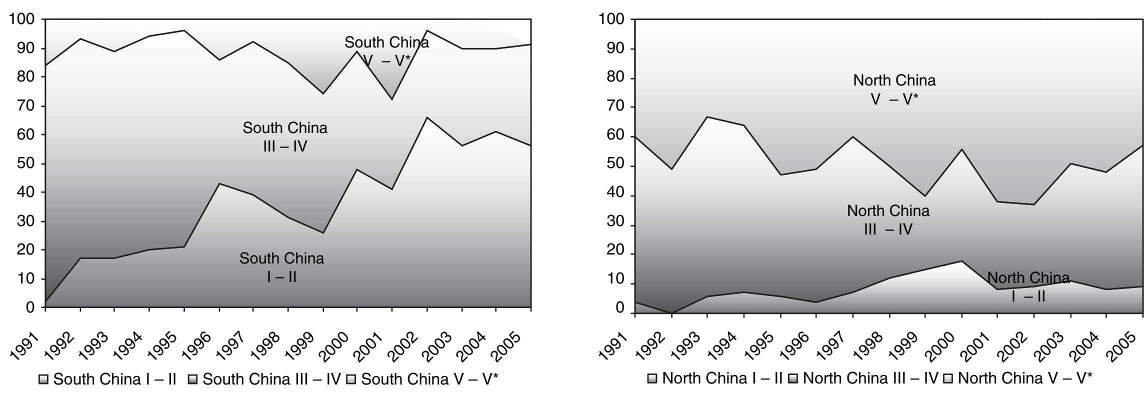
The overall trend for the period 1990 to 2005 indicates that water quality has become substantially better in the water-rich south, but has not improved and may even have worsened in the water-scarce north (see figure 1.5).

In 2004, about 25,000 km of Chinese rivers failed to meet the water quality standards for aquatic life and about 90 percent of the sections of rivers around urban areas were seriously polluted (MWR 2005). Many of the most polluted rivers have been void of fish for many years. Among the 412 sections of the seven major rivers monitored in 2004, 42 percent met the Grade I–III surface water quality standard (that is, water that is safe for human consumption), 30 percent met Grade IV–V standards, and 28 percent failed to meet Grade V. Figure 3.2 (chapter 3) shows for 2004 the location of monitoring stations that failed to meet Class I to III standards. The bulk of the violations occurred in the north in areas of high population density.

Pollution of sea water and lakes is also serious. Thirty percent of sites at which sea water quality is monitored have quality poorer than Grade III. Seventy-five percent of the lakes in China exhibit some degree of eutrophication. Among the 27 major lakes and reservoirs monitored in 2004, none met the Grade I water quality standard, only two (7.5 percent) met the Grade II water quality standard, and five (18.5 percent) met the Grade III quality standard. Most sites have lower quality levels: four (14.8 percent) are Grade IV quality, six (22.2 percent) are Grade V, and ten (37.0 percent) failed to meet the Grade V quality standard. The "Three Lakes" (Taihu, Chaohu, and Dianchi) were among the lakes failing to meet the Grade V water quality standard; total nitrogen and phosphorus were the main pollution indicators contributing to poor water quality (SEPA 2004).

From a health perspective, it is drinking water quality that matters more than surface water quality. Although the last major, nationwide survey of drinking water quality in China occurred in the 1980s, monitoring of drinking water and the sources of drinking water in 300 rural counties, together with data on disease incidence, suggest that polluted drinking water continues to be a problem in rural areas. Due to inadequate treat-

FIGURE 1.5 Average Water Quality in Southern and Northern Rivers, 1991–2005



Source: China Water Quality Management—Policy and Institutional Considerations (World Bank 2006).

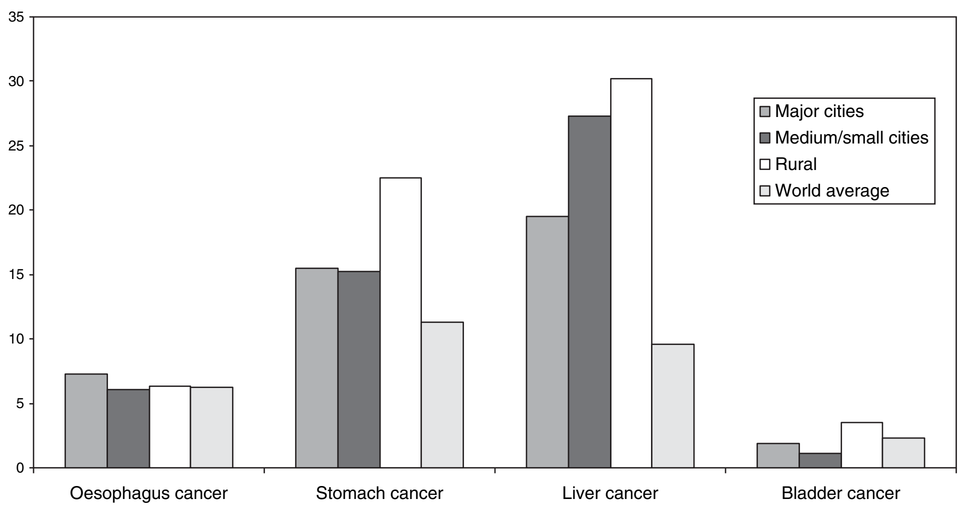
ment, drinking water standards are often violated even in piped water in townships and villages across China. Concerning non-piped water, monitoring data from rural areas show extremely large violations of guidelines. The main problem is land-based contamination. Approximately two-fifths of the rural population does not have piped drinking water, according to the 2005 *China Health Yearbook*. Analyses presented in Chapter 3 of this report suggest a correlation between levels of bacteria and total coliform in drinking water and absence of piped water, as well as a clear relationship between lack of access to piped water and prevalence of diarrhea in children. When it comes to infectious diseases associated with drinking water pollution, however, the annual incidence rates have shown a marked downward trend in the last 20 years.

Although information is not readily available on the percent of the population exposed to various levels of chemical and inorganic pollutants, mortality rates associated with cancers of the digestive system (stomach, liver, and bladder cancers) in rural areas in China suggest that drinking water pollution may still be a serious problem. Figure 1.6 contrasts mortality rates from esophageal, stomach liver, and bladder cancers in different parts of China with world averages. Death rates due to stomach, liver, and bladder cancers in rural China are considerably higher than world averages and also much higher than in large cities in China.

Energy use, industrialization, and urbanization affect environmental performance

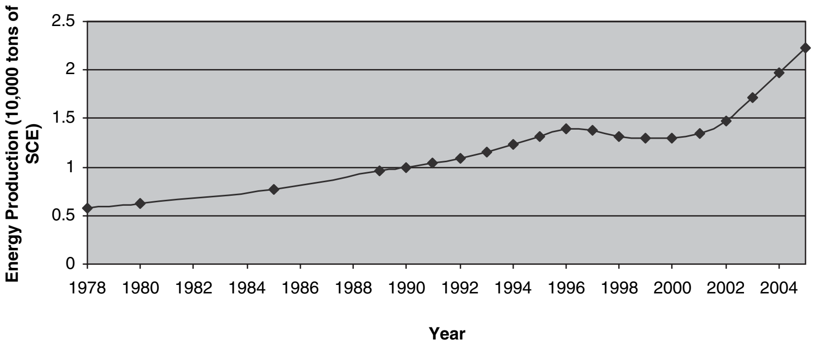
Trends in energy use offer a possible explanation for the recent increase in SO_2 emissions described above. Following the economic slowdown in the late 1990s, the economy grew by about 9 percent each year. Total energy consumption in China increased by 70 percent between 2000 and 2005 (see figure 1.7). Coal consumption accounted for 75 percent of this increase, while the fraction of energy consumption met by hydropower decreased during the 2001–05 period. Moreover, following a marked decrease in the energy intensity of GDP between 1978 and 2001—measured in standard coal equivalents (SCE) used to

FIGURE 1.6 Mortality Rates for Diseases Associated with Water Pollution (per 100,000) in China in 2003 and World Averages in 2000



Source: MoH 2004 and WHO 2006.

FIGURE 1.7 Total Energy Consumption in China, 1978–2005



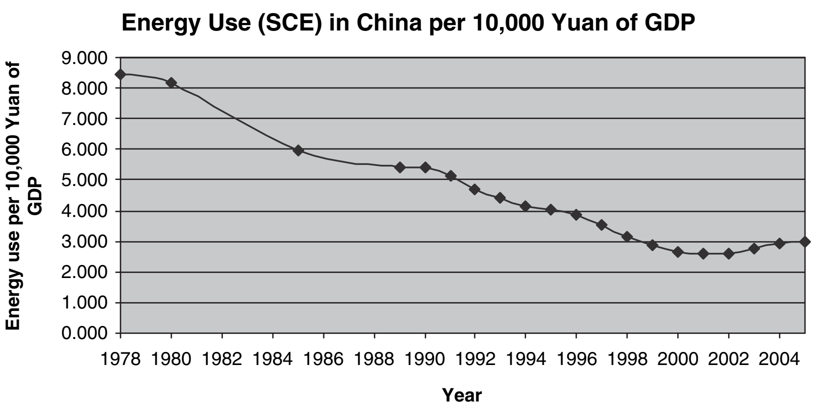
Source: Calculations based upon China Statistical Yearbooks, Various Years.

produce 10,000 Yuan GDP—energy intensity increased in the 2002–05 period (see figure 1.8).

Production of 10,000 Yuan GDP in 1978 required energy equal to 8.43 tons SCE. This was reduced to 2.58 tons in 2001—a 3.2-fold reduction. However, energy intensity increased to 2.76 tons in 2005.

China has also experienced an unprecedented increase in the rate of urbanization. From 2000 to 2005, China's urban population increased by 103 million (see table 1.5). This has likely contributed to increases in urban COD and ammonia nitrogen loads. Although the rate of urban water treatment is increasing (up to 45 percent in 2005), the absolute number of urban residents not linked to water treatment systems has also increased. Moreover, the share of the industries that contribute most to water pollution loads—pulp and paper, food production & processing, textiles, and mining and tanning—have all retained their respective Gross Industrial Output Value (GIOV) in the industrial process. This

FIGURE 1.8 Energy Use (SCE) to Produce 10,000 Yuan of GDP



Source: Calculations based upon China Statistical Yearbooks, Various Years.

TABLE 1.5 China's Urbanization and Industrialization

Year	Total Population	Urban Population (million)	% Urban Population	GIOV Values (Bio RMB in constant 1990 prices)	GIOV Values (indexed)
1978	963	172	18	255	100
1985	1,059	251	24	502	197
1990	1,143	302	26	686	269
1995	1,211	352	29	1723	675
2000	1,267	459	36	2753	1071
2004	1,300	543	42	4083	1600
2005	1,308	562	43	4594	1800

Source: Calculations based upon China Statistical Yearbook various years.

implies that China has yet to realize a substantial reduction in industry-based water pollution due to changes in industrial structure favoring cleaner downstream production.

WATER SCARCITY AND THE USE OF POLLUTED WATER FOR IRRIGATION

Generally speaking, China's water resources are most abundant in the southern and western regions of the country and scarce in the north. The northeast plain areas account for one-third of GDP, but only 7.7 percent of national water resources, while the southwestern areas account for 21.3 percent of national water resources, but only 8.7 percent of GDP.

To cope with water scarcity, China has developed strategies that have to some degree put pressure on the environment. There are three ways that water scarcity harms the environment. First, water scarcity may lead to depletion of groundwater. In some areas of China, the groundwater table has fallen 50 meters since 1960, and it continues to fall 3 to 5 meters annually. Second, water scarcity may lead to excessive consumption of unsafe, polluted water. Consumption of unsafe water in China runs to billions of cubic meters every year. As a third consequence, water scarcity may lead to industry, agriculture, and households being periodically rationed.

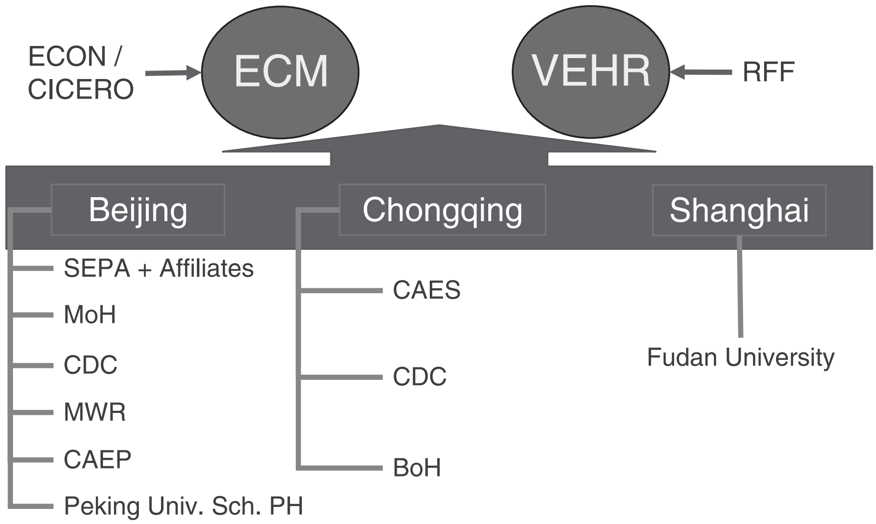
Water depletion and consumption of unsafe water are linked responses to water scarcity. In some areas of China, authorities do not supply unsafe water, with the implication that groundwater depletion increases. For example, this happens in the lower reaches of the Yangtze. It is estimated that 25 billion cubic meters of nonrechargeable deep-aquifer groundwater were mined in China in 2000, 90 per cent of which was used for agricultural purposes.

In other areas, polluted water is used to the maximum extent and water depletion is less than it would have been otherwise. Wastewater irrigation zones are spreading in China and now account for about 4 million hectares of agricultural land. The produce is likely to contain heavy metals such as mercury, cadmium, lead, copper, chromium, and arsenic.

The Chinese Environmental Pollution Impact Model

This report represents the culmination of a joint effort between the Chinese government and a team of Chinese and international experts to assess the costs of environmental degradation in China. The team (see figure 1.9) consisted of staff members from China's State Environmental Protection Administration (SEPA) and affiliates—the Chinese Academy for Environmental Planning, the Policy Research Center of

FIGURE 1.9 Institutions Involved in the Project



Environment and Economy, and the China National Environment Monitoring Center—as well as other government agencies such as the Ministry of Water Resources (MWR), Ministry of Health (MoH), and the Center for Disease Control and Prevention (CDC). The team also included staff from the World Bank, Resources for the Future (USA), CICERO (Norway), and ECON (Norway). It was formed with the intention of both assessing current environmental damages from air and water pollution and developing the tools that would enable these damages to be calculated on a continuing basis at both the national and provincial levels.

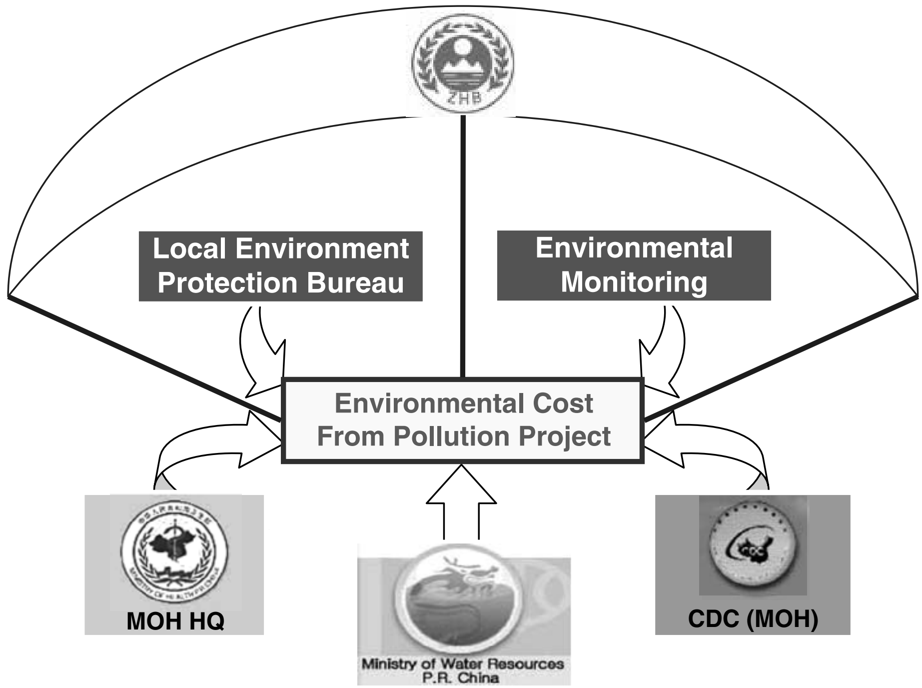
The project, supported by the World Bank, adopted a multi-sectoral approach to assessing the magnitude of air and water pollution in China, with critical data and inputs from SEPA (and its affiliates) and affiliates under the MWR and MoH including CDC).

As part of the multiyear effort to refine methodologies and estimate the costs of pollution, an environmental cost model was developed to (a) help monitor annual environmental impacts; (b) contribute to the development of a National Environmental Accounting System; and (c) contribute to provincial comparisons of environmental performance.

To accomplish these aims, the project was designed to fulfill a set of technical objectives:

1. To formulate, based on Chinese as well as international studies, a Chinese Environmental Cost Model (CECM) that would calculate the damages associated with air and water pollution, by pollutant, sector, and province.
2. To undertake pilot studies on the valuation of health risk (VEHR) that would estimate willingness to pay (WTP) for reductions in premature mortality for use in the CECM.
3. As an integrated part of the CECM, to develop a software tool that would standardize and make operational the calculation of environmental costs.
4. To build capacity for environmental cost calculation in China through collaboration between China's national expert team and an international expert team.

FIGURE 1.10 Main Government Partners in the Project



5. To identify gaps in knowledge—both gaps in research and in the collection of environmental data—that must be filled if the ECM is to form a basis for decision making in China.

It should be emphasized that the outputs of the project can be used for three purposes: (1) to calculate the total damages associated with air and water pollution; (2) as an input to China's Green National Accounts; and (3) to calculate the benefits of programs to reduce air and water pollution. Box 1.1 summarizes how similar analyses have been used in other countries.

This report summarizes the results of the environmental cost model (ECM) and valuation of environmental health risks (VEHR) studies and also describes the methods, data, and literature that have been used to calculate environmental costs in this project. The development of an ECM for China has been aided by three factors:

- The advancement of methods for assessing environmental costs over the past 20 years. Methods to calculate the burden of disease attributable to air and water pollution have advanced significantly, as have methods of estimating the economic costs of environmental degradation.
- The expansion of studies of pollution damages—for example, of the health effects of air pollution—by Chinese researchers. Previous studies of environmental damage in China (World Bank 1997; Cohen et al. 2004) have relied largely on transferring dose-response functions from the international literature to China. A hallmark of the current project is its reliance on studies con-

BOX 1.1 Environmental Cost Models: International Experience

The goal of this project—to quantify environmental degradation using a damage function approach—parallels efforts undertaken by international agencies and governments throughout the world. This box summarizes these efforts.

Global burden of disease due to environmental factors. The World Health Organization (WHO) has calculated (by region) mortality and morbidity associated with both indoor and outdoor air pollution using the same methods as this study. In the case of outdoor air pollution, WHO has estimated annual average PM_{10} concentrations for over 3,000 cities around the world and has used concentration-response functions from Pope et al. (2002) to translate these into premature deaths associated with air pollution. These are calculated by comparing current annual average PM_{10} levels in each city with a reference level of 15 μg/m^3, the same reference level used in the CECM. To calculate the burden of disease associated with indoor air pollution (which is the focus of a separate study), odds ratios from the international literature were applied to the relevant populations exposed to biomass fuels. WHO converts cases of illness and premature mortality into disability-adjusted life-years-saved (DALYs) rather than monetizing cases of illness and premature death.

Benefit-cost analyses of environmental regulations. The United States, United Kingdom, and other members of the European Union regularly conduct benefit-cost analyses of environmental regulations. The techniques used in this report to calculate the health impacts of reducing pollution from current levels to background concentrations—the approach used in calculating the global burden of disease—can also be used to calculate the benefits of smaller reductions in air pollution that are likely to be delivered by various pollution control programs. In the United States (and the EU), the methods described in Chapter 5 of this report are used to monetize health benefits and compare them to costs.

In the United States, benefit-cost analyses must be conducted for all "economically significant" regulations (those costing more than $100 million per year), and are routinely conducted for air quality regulations, following the same protocols used in Chapters 2 and 4 of this report. Benefit-cost analysis is typically used to judge the acceptability of a regulation (do benefits exceed costs?) and sometimes to rank regulatory options—for example, different maximum contaminant levels for arsenic in drinking water (USEPA 2000).

ducted in China, studies that are more appropriate to the Chinese context.

- The improvement in monitoring and environmental data collection in China. Improvements in monitoring of air and water pollution have made it possible to quantify exposures to environmental pollution and estimate associated damages.

Project Components

Pollution costs are typically classified by pollution medium and by the sector affected. Pollution media include air, surface water, drinking water, land-based pollution (solid waste), as well as noise and heat. Pollution damages are usually classified according to their effects—on human health, agriculture, forests, fisheries, or materials (including buildings and monuments). Air pollution or pollution of rivers and lakes may also detract from recreation and aesthetic experiences. The CECM focuses on air and water pollution—both surface and drinking water pollution—but does not include solid waste pollution or radiation at this time. The main sectors for which damages are estimated are health, agriculture, forests, fisheries, materials, and water resources.

In the case of air pollution, the model focuses on particulate matter (TSP or PM_{10}), sulfur dioxide (SO_2), and acid rain. China is the world's largest producer and consumer of coal, much of which has high sulfur content. PM_{10} and SO_2 from coal burning, with attendant acid rain, have caused severe pollution problems in China for decades. Particulate matter is the key air pollutant that has been studied in relation to human health. Associations have been documented between PM and premature mortality; incidence of chronic bronchitis, heart attack, and stroke; respiratory and cardiovascular hospital admissions; and restricted activity days. Acid rain, caused by SO_2 reacting in the atmosphere with water, oxygen, and other substances, can reduce crop and timber yields and forest canopy and damage buildings and monuments, as can SO_2 in gaseous form.

In the case of water pollution, a variety of pollutants are monitored in China, both in surface and drinking water. These include biological pollutants such as coliform bacteria, which are associated with fecal contamination, and chemical pollutants, including naturally occurring elements such as arsenic and fluoride, heavy metals (such as mercury), ammonia, nitrates, and toxic petroleum compounds. From a health perspective, it is drinking water quality that matters most. Epidemiological studies have linked virtually all of the drinking water pollutants in Appendix 2 to either chronic or acute health effects. Eventually, the goal of the CECM is to link specific drinking water pollutants to health endpoints such as cancers of the liver and digestive system; to other chronic diseases, such as diabetes and cardiovascular disease, which have been associated with arsenic; as well as to acute illnesses, such as hepatitis A and dysentery. Another goal is to link surface water pollution to impacts on fish populations and to agriculture. The use of polluted surface water for irrigation reduces both the quantity of agricultural output that is suitable for human consumption and the quality of output. Pollution of surface water may also increase pressure on groundwater resources, contributing to the problem of water scarcity.

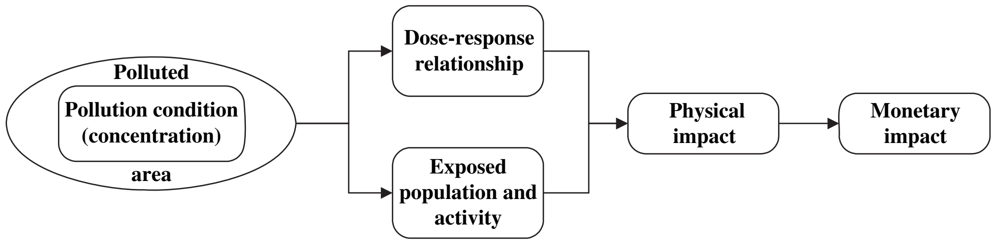
The goal of the CECM is to quantify and, where possible, to monetize the effects of air and water pollution listed in Table 1.6. using a damage function approach. This entails five steps: (1) identifying the nature of the pollution problem—for example, high annual average PM_{10} concentrations in the ambient air or concentration of arsenic in drinking water; (2) identifying the specific endpoints affected (cardiovascular mortality in the case of PM_{10}, or liver cancer in the case of arsenic) and estimating an exposure-response function that links exposure to each endpoint; (3) estimating population exposures (numbers of persons exposed to various PM_{10} concentrations or concentrations of arsenic in drinking water); (4) calculating the physical effects of exposure (deaths due to PM_{10} exposure or cases of liver cancer attributed to arsenic exposure); and (5) assigning a monetary value to the physical effects.

TABLE 1.6 Sectors and Pollutants Included in the CECM

Environmental Sectors	Health	Agriculture	Materials	Forestry	Water Resources	Fishery
Pollutants						
Air pollutants						
TSP (PM_{10})	✓					
SO_2		✓	✓	✓		
Acid rain		✓	✓	✓		
Water pollutants						
In drinking water	✓					
In surface water		✓			✓	✓

Source: the project team.

FIGURE 1.11 Flow Chart for Estimating the Economic Cost of Pollution



Source: the project team.

Step 1: Identify the pollution factors, polluted area, and related conditions.

Step 2: Determine affected endpoints and establish dose-response relationships for pollution damage.

Step 3: Estimate population (or other) exposures in polluted areas.

Step 4: Estimate physical impacts from pollution using information from steps 2 and 3.

Step 5: Convert pollution impacts in physical terms to pollution costs in monetary terms.

The measurement of physical effects attributable to pollution depends crucially on the existence of dose-response functions linking pollution exposure to physical effects, and also on the ability to characterize exposures. This has been done more successfully in the case of human health and air pollution and material damage and air pollution than in other areas. For material damage, exposure-response functions are available for most building materials. However, a comprehensive exposure assessment is more difficult due to lack of data on the amount and surface area of materials in use. Concerning human health, the availability of dose-response functions and data on exposure differ greatly among pollutants and health endpoints. For example, it is much easier to estimate the health effects of PM_{10} in urban areas than to estimate the effects of chronic exposure to arsenic in drinking water.

In China, PM_{10}/TSP and SO_2 are regularly monitored in 341 cities, some of which also monitor nitrogen oxides (NOx). Dose-response functions linking these pollutants to a variety of health outcomes have been estimated by Chinese and international researchers. As a result, estimating the health impacts of air pollution in urban areas is relatively straightforward, at least for acute health effects. In the case of arsenic or other pollutants in drinking water, monitoring data are more difficult to obtain, and the definition of an exposure metric is more complicated than for air pollution.

Drinking water is monitored in a sample of counties by the Chinese Center for Disease Control and Prevention in Beijing, but the samples are not sufficient to obtain an accurate estimate of the fraction of the population exposed to different concentrations of pollutants in their drinking water throughout the country. Thus, although there are epidemiologic studies linking arsenic to liver cancer, it is difficult to apply them, as indicated in Figure 1.11, for lack of exposure data.

The absence of dose-response functions becomes more of a problem when examining the effects of pollution in non-human populations. For example, the literature linking fish populations to surface water pollution (either to acid rain, or to eutrophication of lakes due to nitrogen loadings) is sparse. So is the literature linking acid rain or SO_2 to timber yields and to tree cover. This makes it difficult—in China, but also in Western countries—to quantify the effects of air and water pollution on forests and fisheries. For these reasons, it has not been possible to quantify all of the effects checked in Table 1.6.

The remainder of this report summarizes the current state of analysis of the effects of air and water pollution in the CECM. It is divided into 6 chapters, organized as follows:

Chapter 2. The Health Impacts of Ambient Air Pollution. The CECM quantifies cases of chronic bronchitis, premature mortality, and respiratory and cardiovascular hospital admissions associated with PM_{10} in urban areas in China. This is a bottom-up analysis, conducted at the city level, and aggregated to the provincial and national levels. A distinguishing feature of the CECM is its use of Chinese concentration-response functions rather than relying solely on dose-response transfer from the international literature.

Chapter 3. The Health Impacts of Water Pollution. As noted above, it is not possible to measure population exposures to the pollutants listed in Table 1.6 from available data. This chapter presents an overview of surface water pollution in China, as well as information on the source of drinking water and the nature of drinking water treatment. Information on the levels of specific pollutants in drinking water is presented for a sample of rural counties, as well as for selected districts in Chongqing. Information on the incidence of diseases that have been associated with various drinking water pollutants is presented, together with a disease matrix summarizing associations found in the Chinese and international literature. An attempt is made to compute the impact of polluted drinking water on cancer incidence in rural areas. The chapter concludes with original research linking incidence of diarrheal disease among children under 5 living in rural areas of China with availability of piped water.

Chapter 4. Valuing Environmental Health Effects. An important goal of the CECM/VEHR project is to contribute to the literature on health valuation in China. This chapter summarizes the results of original studies conducted in Shanghai and Chongqing to estimate people's willingness to pay to reduce risk of premature death. The chapter also discusses the Adjusted Human Capital (AHC) approach—the official approach used to value health costs in China, and uses both approaches to value premature mortality associated with air pollution. Estimates of the value of air-pollution-related morbidity are also presented, as well as the health impacts of water pollution.

Chapter 5. The Non-Health Impacts of Water Pollution. This chapter concentrates on the impacts from water pollution, where pollution of surface water bodies can reduce agricultural yields and harvests of fish. It estimates the damages associated with acute pollution incidents affecting fisheries and the damages associated with the use of sewage-contaminated water for irrigation of crops. It also deals with the related issue of water scarcity caused by pollution.

Chapter 6. The Non-Health Impacts of Air Pollution. This chapter focuses on the non-health impacts from air pollution, including SO_2 and acid rain damage to buildings and other materials and their impacts on crop and timber yields. It values damages to buildings in South China and crop losses due to acid rain and SO_2 pollution throughout the country using Chinese dose-response information. Effects on forests are not quantified due to lack of data on the area planted, by species, and lack of appropriate dose-response functions.

Table 1.7 below highlights some important types of environmental damages that were not quantified due to lack of sufficient data.

TABLE 1.7 Environmental Damages in the CECM

Quantified Damages	Non-quantified Damages	Why Not Quantified
Health effects of ambient PM_{10}	Health effects of ambient ozone	1
Diarrheal disease associated with no piped water connection; cancers associated with water pollution	Health effects associated with chemical and inorganic water pollutants	1
SO_2 and acid rain damage to crops	SO_2 and acid rain damage to forests	1,2
SO_2 and acid rain damage to buildings	SO_2 and acid rain damage to other types of construction	1
Acute effects of water pollution on fish	Chronic effects of water pollution on fish	1,2
Agricultural damages from wastewater irrigation		

1 = Effect not quantified due to insufficient information about exposure
2 = Effect not quantified due to insufficient information about dose-response
Source: the project team.

ANNEX 1. Concentration Values of Pollutants in Ambient Air

Name of Pollutant	Time	Concentration Values			Concentration Level Unit
		Class I	Class 2	Class 3	
SO_2	Yearly average	0.02	0.06	0.10	
	Daily average	0.05	0.15	0.25	
	Hourly average	0.15	0.50	0.70	
TSP	Yearly average	0.08	0.20	0.30	Mg/m^3
	Daily average	0.12	0.30	0.50	
PM_{10}	Yearly average	0.04	0.10	0.15	
	Daily average	0.05	0.15	0.25	
NO_x	Yearly average	0.05	0.05	0.10	
	Daily average	0.10	0.10	0.15	
	Hourly average	0.15	0.15	0.30	
NO_2	Yearly average	0.04	0.04	0.08	
	Daily average	0.08	0.08	0.12	
	Hourly average	0.12	0.12	0.24	
CO	Daily average	4.00	4.00	6.00	
	Hourly average	10.00	10.00	20.00	
O_3	Hourly average	0.12	0.15	0.20	
Pb	Seasonal average		1.50		
	Yearly average		1.00		$\mu g/m^3$
B(a)P	Daily average		0.01		
	Daily average		7[a]		
F	Hourly average		20[a]		
	Monthly average	1.8[b]		3.0[c]	$\mu g/dm^2$. d?
		1.2[b]		2.0[c]	

a. Urban area
b. Pasturing area, or Part Farming—Part Pasturing, or Silkworm-mulberry producing area
c. Farming and Forestry Area

ANNEX 2. List of Pollutants Monitored in Surface Water and Their Standards (mg/L)

No.	Parameters		Categories I	II	III	IV	V
	Basic requirements		All water bodies should not contain substances from non-natural causes as listed below: a. Any substance that can subside and form offensive sediments b. Floating matter, such as fragments, floating scum, oils, or any other materials that can offend sense organs c. Any substance that produces offensive color, odor, taste, or turbidity d. Any substance that can harm human beings, animals, and plants, or cause toxic or adverse physiological reactions e. Any substance that can easily cause the breeding of offensive aquatic organisms				
1	Water temperature (C°)		Temperature changes in the water environment induced by human activities should be within: Summer weekly average maximum temperature rise ≤ 1 Winter weekly average maximum temperature down ≤ 2				
2	pH		6.5 ~ 8.5 (mg/L)				6 ~ 9 (mg/L)
3	Sulfate (as SO_4^{2-})*	≤	below 250	250	250	250	250
4	Chloride (as Cl^-)*	≤	below 250	250	250	250	250
5	Soluble iron*	≤	below 0.3	0.3	0.5	0.5	1.0
6	Total manganese*	≤	below 0.1	0.1	0.1	0.5	1.0
7	Total copper*	≤	Below 0.01	1.0 (0.01 for fishery)	1.0 (0.01 for fishery)	1.0	1.0
8	Total zinc*	≤	0.05	1.0 (0.1 for fishery)	1.0 (0.1 for fishery)	2.0	2.0
9	Nitrate (as N)	≤	Below 10	10	20	20	25
10	Nitrate (as N)	≤	0.06	0.1	0.15	1.0	1.0
11	Non-ionic ammonia	≤	0.02	0.02	0.02	0.02	0.02
12	Kjeldahl nitrogen	≤	0.5	0.5	1.0	2.0	2.0
13	Total phosphorus (as P)	≤	0.02	0.1 (0.025 for reservoirs and lakes)	0.1 (0.05 for reservoirs and lakes)	0.2	0.2
14	Permanganate value	≤	2.0	4.0	6.0	8.0	10.0
15	Dissolved oxygen	≤	90% of saturation value	6.0	5.0	3.0	2.0
16	Chemical oxygen Demand (COD_{Cr})	≤	Below 15	Below 15	15	20	25
17	Biological oxygen Demand (BOD_5)	≤	Below 3.0	3.0	4.0	6.0	10
18	Fluoride (as F^-)	≤	Below 1.0	1.0	1.0	1.5	1.5
19	Selenium (IV)	≤	Below 0.01	0.01	0.01	0.02	0.02
20	Total arsenic	≤	0.05	0.05	0.05	0.1	0.1
21	Total mercury**	≤	0.00005	0.00005	0.00001	0.001	0.001
22	Total cadmium***	≤	0.001	0.005	0.005	0.005	0.01
23	Chromium (VI I)	≤	0.01	0.05	0.05	0.05	0.1
24	Total lead**	≤	0.01	0.05	0.05	0.05	0.1
25	Total cyanide	≤	0.0005	0.05 (0.005 for fishery)	0.02 (0.005 for fishery)	0.2	0.2
26	Volatile phenol**	≤	0.002	0.002	0.005	0.01	0.1
27	Oils** (Petroleum ether extraction)	≤	0.05	0.05	0.05	0.5	1.0
28	Anionic surfactant	≤	Below 0.2	0.2	0.2	0.3	0.3
29	Coli-index*** (Individuals/L)	≤			10000		
30	Benzo (a) pyrene (pg/L)	≤	0.0025	0.0025	0.0025		

ANNEX 3. Pollutants Monitored in Drinking Water in China and Drinking Water Standards

Drinking Water Pollutants	Class I	Class II	Class III
Chrome (degree)	15.0	20.0	30.0
Turbidity (degree)	3.0	10.0	20.0
Total dissolved solids (mg/L, $CaCO_3$)	450.0	550.0	700.0
Iron (mg/L)	0.3	0.5	1.0
Manganese (mg/L)	0.1	0.3	0.5
COD (mg/L)	3.0	6.0	6.0
Chlorate (mg/L)	250.0	300.0	450.0
Sulfate (mg/L)	250.0	300.0	400.0
Fluoride (mg/L)	1.0	1.2	1.5
Arsenic (mg/L)	0.1	0.1	0.1
Nitrate (mg/L)	20.0	20.0	20.0
Total bacteria (/mL)	100.0	200.0	500.0
Total coliform (/L)	3.0	11.0	27.0

2

Health Impacts of Ambient Air Pollution

This chapter reviews the health effects associated with particulate matter, summarizes population exposure to PM_{10} in China and describes the techniques used to estimate the health damages associated with PM_{10} exposure in 2003. Specifically, the CECM quantifies cases of chronic bronchitis, premature mortality, and respiratory and cardiovascular hospital admissions associated with PM_{10} in urban areas in China.

This is a bottom-up analysis, conducted at the city level, and aggregated to the provincial and national levels. A distinguishing feature of the CECM is its use of Chinese concentration-response functions rather than relying solely on dose-response transfer from the international literature. The premature deaths and cases of illness quantified using the techniques described in this chapter are valued in Chapter 4.

Energy consumption, especially coal consumption, is the main source of air pollutants such as particles, SO_2, NO_x, and CO in most cities of China. As the primary energy source, coal has accounted for about 65 to 70 percent (China Statistical Yearbook 2004) of total energy consumption in recent years, which has caused many environmental and human health problems. Crude oil consumption has been increasing because of the rapid expansion of the motor vehicle fleet in many cities. In recent years, epidemiological studies conducted around the world have demonstrated that there are close associations between air pollution and health outcomes. PM_{10} and SO_2 are chosen in many studies as the indicative pollutants for evaluating the health effects of ambient air pollution. Although the mechanisms are not fully understood, epidemiological evidence suggests that outdoor air pollution is a contributing cause of morbidity and mortality. Epidemiological studies have found consistent and coherent associations between air pollution and various outcomes, including respiratory symptoms, reduced lung function, chronic bronchitis, and mortality.

In China, epidemiological studies have been conducted beginning in the 1980s and 1990s in Beijing, Shenyang, Shanghai, and other cities. These include two time-series analyses of the relationship between daily air pollution and hospital outpatient visits/emergency room visits and daily cause-specific population mortality in urban areas of Beijing (Chang et al. 2003; Chang, Wang, and Pan 2003), a meta analysis of exposure-response functions between air pollutants and cause-specific mortality derived from Chinese studies, and a regression analysis of environmental monitoring data and population mortality data for over 30 cities of China. (See CD-ROM A.1). These study results suggest that urban air pollution in China causes significant public health impacts and economic damage to the exposed populations. They provide a good foundation for further evaluation of the health impacts of air pollution in China.

HEALTH OUTCOMES FROM AIR POLLUTION

Epidemiological research has found consistent and coherent associations between air pollution and various health endpoints, or health effects. These include reduced lung function, respiratory symptoms, chronic bronchitis, cardiovascular and cerebrovascular diseases, hospitalization, outpatient visits, work and school absenteeism, and premature death. Although the mechanisms are still not fully understood, research during the past 10 to 20 years suggests that outdoor air pollution contributes to morbidity and mortality linked with respiratory, cardiovascular, and cerebrovascular illness and diseases. Some effects may arise from short-term exposure, while others are associated with long-term exposure. When we select health endpoints to be accounted for in the environmental cost model, the basic principles are as follows:

- First priorities should be given to the health endpoints that are registered in Chinese cities on a regular basis and classified by ICD-9 code (or by ICD-10, the latest revision of the classification system). This will ensure data availability and enable comparisons between regions. These data include population mortality, hospital admissions, and hospital outpatient/emergency visits.
- There are well-documented studies of exposure-response functions between concentrations of air pollutants (exposure) and the given health endpoints (response).
- The methodologies applied in the epidemiological studies forming the basis for exposure-response functions should be as similar as possible to studies in other countries to facilitate comparison.

As noted above, the selection of health endpoints is restricted by the availability of exposure-response studies. In this assessment we have selected all-cause mortality, hospital admissions for respiratory and cardiovascular disease, and incidence of chronic bronchitis as endpoints because of the availability of exposure-response functions. Health endpoints can be classified in broad disease groups or specified in detail according to ICD codes. Different studies on exposure-response relationships may address more or less specific health endpoints. Typically, studies report steeper exposure-response coefficients when cause-specific health endpoints are addressed, as opposed to studies focusing on broader groups of endpoints. For these endpoints, we therefore have to apply a relatively crude classification, which increases the uncertainty of the results.

In health cost estimation, it is also important to make sure that the endpoints in the exposure-response functions are consistent with the endpoints for which statistical data are available. At present, the health data from regular surveillance is often insufficient, and the system for reporting prevalence of morbidity is not complete, especially for some chronic diseases. This limits the choice of health endpoints. Since there is no requirement from the Ministry of Health for cause-specific registration for emergency visits (EVs) and outpatient visits (OPVs), OPVs and EVs for respiratory and cardiovascular diseases cannot enter endpoint lists despite documented studies on their dose-response coefficients.

In line with the above principles, the health endpoints evaluated in this project are described as follows:

- *Mortality.* all-cause mortality
- *Morbidity.* respiratory and cardiovascular hospital admissions; incidence of chronic bronchitis

Two endpoints related to hospitalization are selected, covering the bulk of hospital admissions attributable to air pollution. The two endpoints are hospital admissions due to cardiovascular diseases and hospital admissions due to respiratory diseases. A broad range of diagnoses, specified by their ICD-9 code, were included in the studies from which the exposure-response functions are derived, including cerebrovascular diseases, pulmonary heart diseases, ischaemic heart disease, COPD, and pneumonia. The prevalence of chronic respiratory symptoms and diseases in a population is related to long-term, integrated exposure. Prevalence rates are often higher in adults than in children. We selected the endpoint chronic bronchitis as an endpoint presumably representing an important share of the economic impact and human suffering associated with air pollution. Chronic bronchitis typically constitutes the largest share of cases of chronic obstructive pulmonary diseases (COPD) and covers a range of sub-diagnoses, which are all likely to entail substantial reduced well-being and restricted activity.

CAUSAL AGENTS AND THRESHOLD VALUES

Causal Agents in Air-Pollution-Related Disease

Although adverse effects on human health from particulate matter, SO_2, O_3, NOx, and CO are documented, most studies have focused on the relationship between SO_2, particulate matter, and respiratory and cardiovascular diseases. After thorough consideration, we decided to choose PM_{10} as the single air pollutant index for the following reasons:

1) Ambient SO_2 concentrations in most Chinese cities have greatly decreased compared with a few years ago, and are in many cities now lower than the WHO *Air Quality Guideline* (2000) of 50μg/m^3. The air quality monitoring results from Chinese cities in 2003 showed that, among the 341 monitored cities, the annual average ambient SO_2 concentration exceeded the Class-II standard (60μg/m^3) in 26 percent of the cities. Fifty-five percent of the cities had annual average PM_{10} (TSP) levels violating the Class-II standard (100μg/m^3). Annual average NO_2 concentrations of all monitored cities met the Class-II standard (50μg/m^3). This suggests that particulate matter has become the air pollutant of primary concern in China.
2) Different air pollutants may have a synergetic effect on human health. For instance, the combined effect of SO_2 and PM_{10} may be higher (or lower) than the sum of the two components when they occur in isolation. Moreover, a part of PM_{10} may be sulfate, which is converted SO_2. In spite of a large body of studies, the contribution of each of these pollutants to health damage is difficult to disentangle. In our view, adding the health cost from, respectively, PM_{10} and SO_2 may lead to double counting.
3) The trial calculation results showed that the health cost estimated for SO_2 (based on the dose-response coefficients in the December 2002 *Progress Report of Chinese Environmental Cost Model*) represented only about one-tenth of the total health cost due to air pollution.

Because particulate matter under 10 μm is an important vector for several toxic and hazardous air pollutants and because of the close relationship between PM_{10} and health effects found in many epidemiological studies, we exclude SO_2 from the final estimation to avoid double counting.

Air Pollutant Thresholds

According to WHO (2000), there is no level below which particulate matter may not result in health effects in the susceptible population, but there is a lower limit to the level at which results have been reported in epidemiological studies. We define 15μg/m^3 as the lower threshold value for PM_{10} effects, given that the lowest PM_{10} concentration observed in the ACS cohort study by Pope (1995) is 15μg/m^3. This lower threshold is also applied by WHO (Cohen et al. 2004).

POPULATION EXPOSURE

China is the largest developing country in the world. Of the more than 1.3 billion inhabitants, about 40 percent live in urban areas (China Statistical Yearbook 2004). With the growing economy, many cities in China have to face the challenge of air pollution. The health impacts of air pollution, especially in cities, are gradually being acknowledged by researchers, government, and the public. It is difficult to estimate the number of people exposed to high levels of air pollution at a national level, because there are large variations across different geographic and meteorological areas, as well as across socioeconomic groups. Generally, ambient air pollution is closely associated with industrialization and urbanization. Hence, the urban population is likely to be the primary group exposed to high levels of ambient air pollution. Moreover, the state-controlled air pollution monitoring sites are distributed in urban areas only, so the air quality data represent air pollution levels in cities. "Exposed population" in this health damage valuation refers to urban residents, defined as the population of urban districts as given in the *China City Statistical Yearbook* (2004).

Table 2.1 shows the percentage of the urban population exposed to different classes of PM_{10} levels in the 31 provinces of mainland China. Figure 2.1 maps the percentage of urban population exposed to Class III and > Class III PM_{10} levels. Over half of the urban population in China is exposed to annual average PM_{10} levels greater than or equal to the Class III standard (100 $\mu g/m^3$). Over 11 percent are exposed to PM_{10} levels in excess of 150 $\mu g/m^3$, which is three times the U.S. annual average standard. The provinces with the largest percentage of people exposed to PM_{10} levels greater than or equal to the Class III standard are generally in the north, while eastern and southern provinces with high population densities—Shandong, Guangdong, and Jiangsu—have the highest numbers of people exposed.

There are few air pollution monitoring stations in rural areas in China, so we cannot evaluate the effects of outdoor air pollution on the human health of the population in rural areas. Chapter 3 of this report estimates the health impacts of indoor air pollution in rural areas in China.

EXPOSURE-RESPONSE RELATIONSHIPS

Review of Epidemiological Evidence

The effects of air pollution on human health include the chronic effects of long-term exposure and the acute effects of short-term exposure. In the past two decades, a large number of studies—especially short-term, time-series studies—have reported exposure-response relationships between air pollution exposure and human health. Long-term cohort studies provide the best method to evaluate the chronic effects of air pollution on human health, whereas time-series studies are appropriate for revealing the acute effects of short-term fluctuations in pollution levels. Exposure-response coefficients from cohort studies of premature mortality are typically several times higher than coefficients reported in time-series studies. We assumed that the short-term effects found in time-series studies are embedded in the long-term effects on mortality rates derived from cohort studies.

A large number of time-series studies of mortality have been published in the past 20 years, but only a few cohort studies have appeared. In China, there are some time-series studies and several cross-sectional mortality studies, conducted in cities such as Beijing (Chang et al. 2003; Chang, Wang, and Pan 2003; Dong et al. 1995; Dong et al. 1996; Gao et al. 1993; Xu et al. 1995; Xu et al. 1994), Shanghai (Kan and Chen 2003; Kan and Chen 2004), Shenyang (Wang, Lin, and Pan 2003; Xu et al. 1996a; Xu et al. 2000; Xu et al. 1996b), and Chongqing (Venners et al. 2003).

To derive exposure-response functions for air pollution and mortality applicable to the entire

TABLE 2.1 PM_{10} Pollution Exposure of the Urban Population (population in 10,000's)

		I Class	II Class	III Class	>III Class	
Provinces	**Item**	**PM_{10}< 40μg/m^3**	**PM_{10}: 40–100μg/m^3**	**PM_{10}: 100–150μg/m^3**	**PM_{10} >150μg/m^3**	**Total Population/%**
Beijing	Population	0	0	1,079	0	1,079
	%	0.00	0.00	100.00	0.00	100
Tianjin	Population	0	0	759	0	759
	%	0.00	0.00	100.00	0.00	100
Hebei	Population	0	384	1,650	496	2,529
	%	0.00	15.16	65.23	19.60	100
Shanxi	Population	0	148	322	796	1,267
	%	0.00	11.68	25.45	62.87	100
Neimeng	Population	0	144	311	240	694
	%	0.00	20.67	44.74	34.59	100
Liaoning	Population	0	1,615	1,265	78	2,958
	%	0.00	54.61	42.75	2.64	100
Jilin	Population	0	807	473	407	1,687
	%	0.00	47.80	28.06	24.14	100
Heilong Jiang	Population	0	627	841	179	1,647
	%	0.00	38.08	51.04	10.88	100
Shanghai	Population	0	1,278	0	0	1,278
	%	0.00	100.00	0.00	0.00	100
Jiangsu	Population	0	639	3516	458	4,613
	%	0.00	13.84	76.22	9.94	100
Zhejiang	Population	0	1,532	1,782	0	3,314
	%	0.00	46.22	53.78	0.00	100
Anhui	Population	0	927	1062	0	1,990
	%	0.00	46.61	53.39	0.00	100
Fujian	Population	0	1,243	385	79	1,707
	%	0.00	72.81	22.57	4.62	100
Jiangxi	Population	0	448	800	159	1,407
	%	0.00	31.85	56.85	11.30	100
Shandong	Population	0	3,610	1,546	190	5,345
	%	0.00	67.53	28.92	3.55	100
Henan	Population	0	792	1,706	738	3,236
	%	0.00	24.47	52.72	22.81	100
Hubei	Population	0	1,520	2,351	0	3,871
	%	0.00	39.26	60.74	0.00	100
Hunan	Population	0	317	1,594	358	2,269
	%	0.00	13.97	70.23	15.80	100
Guang Dong	Population	0	5,005	293	0	5,298
	%	0.00	94.47	5.53	0.00	100
Guangxi	Population	204	473	689	254	1,619
	%	12.57	29.20	42.55	15.68	100
Hainan	Population	354	113	0	0	467
	%	75.86	24.14	0.00	0.00	100
Chong Qing	Population	0	0	1,488	0	1,488
	%	0.00	0.00	100.00	0.00	100
Sichuan	Population	0	489	1,337	1,276	3,103
	%	0.00	15.77	43.09	41.14	100
Guizhou	Population	0	357	582	0	939
	%	0.00	38.00	62.00	0.00	100

(*continued*)

TABLE 2.1 PM_{10} Pollution Exposure of the Urban Population (population in 10,000's) (*Continued*)

Provinces	Item	I Class: PM_{10}< 40μg/m³	II Class: PM_{10}: 40–100μg/m³	III Class: PM_{10}: 100–150μg/m³	>III Class: PM_{10} >150μg/m³	Total Population/%
Yunnan	Population	64	789	76	0	929
	%	6.91	84.95	8.14	0.00	100
Xizang	Population	0	14	0	0	14
	%	0.00	100.00	0.00	0.00	100
Shaanxi	Population	0	147	721	340	1,207
	%	0.00	12.19	59.69	28.13	100
Gansu	Population	0	124	339	272	735
	%	0.00	16.92	46.04	37.04	100
Qinghai	Population	0	0	107	12	119
	%	0.00	0.00	89.92	10.08	100
Ningxia	Population	0	0	72	157	229
	%	0.00	0.00	31.34	68.66	100
Xinjiang	Population	0	181	278	226	684
	%	0.00	26.37	40.66	32.96	100
Total	Population	622	23,720	27,422	6,716	58,480
	%	1.06	40.56	46.89	11.48	100

Source: authors calculations.
Note: The PM_{10} pollution exposure is computed based on data from 660 cities. Because air pollution monitoring data in China are available for 341 cities, the air pollution levels of non-monitored county-level cities refer to the data of their upper-level prefecture cities.

country, we undertook a systematic literature review and analyzed the available studies by means of meta-analysis and statistical trend analysis, and made a final selection according to the criteria mentioned above.

Cohort studies of long-term exposure

Cohort studies take advantage of spatial variation in air pollution concentrations to compare the incidence of disease and death in populations exposed over the long term to differing levels of air pollution. By following large populations for many years, cohort studies estimate both numbers of deaths and, more importantly, mean reduction in life span attributable to air pollution.

Evidence from cohort studies of populations in the United States indicates that long-term exposure to outdoor air pollution is associated with an increase in total mortality, cardiopulmonary mortality, and lung cancer mortality in adults. These cohort studies include the Harvard six-city study (Dockery et al. 1993), the ACS cohort study (Pope et al. 1995), and the ACS extended study (Pope et al. 2002). The main background information and results are shown in Tables 2.2 and 2.3.

Ecological studies of air pollution and human health

There is no cohort study in China and only three cross-sectional studies that reflect the effects of long-term air pollution exposure on mortality. In China, Jing et al. (1999), Xu et al. (1996a, 1996b, 2000), and Wang et al. (2003) investigated the chronic effects of air pollution on mortality in Shenyang and Benxi. They estimated relative risks by comparing mortality rates in the

FIGURE 2.1 Urban Population Exposed to Class III and > Class III PM10 Levels, 2003

Source: Based upon Table 2.1

TABLE 2.2 Background of Cohort Studies in the United States

Authors	Year	Locations	Pollutants	Concentration Ranges	Study Design
Dockery et al.	1993	U.S. 6 cities	PM_{10}	18.2~46.5ug/m3	Cohort study
Pope et al.	1995	U.S. 61 cities	$PM_{2.5}$	9.0~33.5ug/m3	Cohort study
Pope et al.	2002	U.S. 61 cities	$PM_{2.5}$	Mean=17.7ug/m3	Cohort study

Sources: Dockery et al. 1993; Pope et al. 1995; Pope et al. 2002.

TABLE 2.3 Main Results of Long-Term Cohort Studies in the U.S.A.

Authors	Health End Points	Pollutants	RR	95% C.I.	Beta	Std Error.
Dockery et al.	All Cause	PM_{10}	1.26	1.08,1.47	0.82	0.28
	Lung Cancer		1.37	0.81,2.31	1.10	0.94
	Cardiopulmonary		1.37	1.11,1.68	1.10	0.37
Pope et al.	All Cause	$PM_{2.5}$	1.17	1.09,1.26	0.64	0.15
	Lung Cancer		1.03	0.80,1.33	0.12	0.53
	Cardiopulmonary		1.31	1.17,1.46	1.10	0.23
Pope et al.	All Cause	$PM_{2.5}$	1.04	1.01,1.08	0.40	0.16
	Lung Cancer		1.08	1.011,1.16	0.79	0.35
	Cardiopulmonary		1.06	1.02,1.11	0.57	0.22

Source: Dockery et al. 1993; Pope et al. 1995; Pope et al. 2002.
Note: In the study by Dockery et al., RR is the mortality-rate ratio for the most polluted of the cities as compared with the least polluted. In the studies by Pope et al., RR is the relative risk associated with a 10 μg/m³ change in particulate pollution. Beta is the percentage increase in health effect per μg/m³ increment of air pollutant.

worst-polluted and the least-polluted areas of each city. The background and main results are shown in Tables 2.4 and 2.5.

Transformation of TSP and $PM_{2.5}$ to PM_{10}

The particulate matter indices, including PM_{10}, $PM_{2.5}$, and TSP, differ in the above cohort studies and ecological studies. In order to be applied in the ECM and compared with each other, we convert to the uniform indicator index—PM_{10}. Aunan and Pan (2004) suggest that the conversion ratio of TSP to PM_{10} is 0.60. In Dockery's six-city study (Dockery et al., 1993), the ratio of $PM_{2.5}$ to PM_{10} is 0.60 to 0.64. Lvovsky et al. (2000) suggest that the ratio is 0.65. In the recent Chinese four- city study (Qian et al., 2001), the ratio is 0.51~0.72. Wan (2005) found an average ratio of 0.55 in 28 cities in China. We apply a conversion ratio of 0.60 for $PM_{2.5}$ to PM_{10} and a ratio of 0.50 for PM_{10} to TSP. The results are shown in Tables 2.6 and Table 2.7.

Time-series Studies of Short-term Exposure and Morbidity

Time-series studies have been conducted to analyze the relationship between daily rates of health events, such as hospital admissions or deaths, and daily concentrations of air pollutants and other risk factors (e.g., weather). In time-series studies, individual factors—such as smoking, nutrition, behavior and genetic characteristics—are unlikely to be confounders because they are generally constant throughout the study period.

TABLE 2.4 Background of Ecological Studies in China

Authors	Year	Locations	Pollutants	Concentration Ranges	Study Design
Jing et al.	1999	Benxi	TSP	290~620ug/m³	Cross-sectional ecological study
Xu et al.	1996	Shenyang	TSP	353~560ug/m³	Cross-sectional ecological study
Wang et al.	2003	Shenyang	TSP	200~540ug/m³	Cross-sectional ecological study

Source: Jing et al. 1999; Xu et al. 1996; Wang et al. 2003.

TABLE 2.5 Main Results of Cross-Sectional Ecological Studies in China

Authors	Health End Points	Pollutants	RR	95% C.I.	Beta	Std Error.
Jing et al.	All Cause	TSP	1.08	1.02,1.14	0.077	0.028
	COPD		1.24	1.04,1.44	0.22	0.083
	CVD		1.24	1.08,1.41	0.22	0.068
	CEVD		1.08	1.00,1.15	0.077	0.036
Xu et al.	All Cause	TSP	1.20	1.15,1.24	0.059	0.0063
	COPD		1.22		0.065	
	CEVD & CVD		1.21		0.062	
	Coronary-heart-disease		1.11		0.034	
Wang et al.	CVD	TSP	1.01	1.00,1.02	0.024	0.0087

Source: Jing et al 1999; Xu et al. 1996; Wang et al. 2003.
Note: Beta is the percentage increase in health effects per 1μg/m3 increment of TSP.

Various regression techniques are used to estimate a coefficient that represents the relationship between exposure to air pollution and human health outcomes. The usual regression methods model the logarithm of the response variable, such as daily deaths or hospital admissions, to estimate the relative risk, or proportional change in the outcome per increment of ambient pollution concentration. Table 2.8 presents the results of meta-analysis of time series morbidity studies conducted in China (Aunan and Pan 2004).

Limitations of Previous Studies

The exposure-response functions mentioned above are based on research conducted in China and other countries during the past 10 years. A range of factors may affect the magnitude of the exposure-response coefficients. These factors may have changed since the older studies were carried out, including the general health status and living conditions of the population, and the level and composition of air pollution. The main limitations of previous studies are the following:

Change of air pollution level and type

Historically, air pollution in urban areas in China has come primarily from coal combustion. Up to 5 to 10 years ago, research was focused on the impacts of SO_2 on human health. In recent years, air pollution in urban areas in China has been transformed from coal-smog air pollution into a mixture of coal-smog and automobile exhaust. Emissions of SO_2 have decreased gradually and particulate matter has become the principal pollutant of concern in most cities in China.

TABLE 2.6 Summary of the Results of Long-Term Exposure Studies (PM_{10})

Authors	Health End Points	Beta	Std Error
Dockery et al. 1993	All Cause	0.82	0.28
	Lung Cancer	1.11	0.94
	Cardiopulmonary	1.11	0.37
Pope et al. 1995	All Cause	0.38	0.09
	Lung Cancer	0.07	0.32
	Cardiopulmonary	0.66	0.14
Pope et al. 2002	All Cause	0.24	0.10
	Lung Cancer	0.47	0.21
	Cardiopulmonary	0.34	0.13
Jing et al. 1999	All Cause	0.15	0.06
	COPD	0.43	0.17
	CVD	0.43	0.14
	CEVD	0.15	0.07
Xu et al. 1996	All Cause	0.12	0.01
	COPD	0.13	
	CEVD	0.12	
	Coronary-heart-disease	0.07	
Wang et al. 2003	CVD	0.041	0.02

Source: Dockery et al. 1993; Pope et al. 1995; Pope et al. 2002; Jing et al. 1999; Xu et al. 1996; Wang et al. 2003.

TABLE 2.7 Exposure-Response Relationship for Long-Term Impact of PM_{10} on Mortality Rates

	Dockery	Pope, 1995	Pope, 2002	Jing et al.	Xu et al.	Wang et al.
	1	2	3	4	5	6
All Cause	0.82	0.38	0.24	0.15	0.12	
Lung Cancer	1.11	0.072	0.47			
Cardiopulmonary	1.11	0.66	0.34			
COPD				0.43	0.13	
CVD				0.43		0.041
CEVD				0.15	0.12	
PM_{10} (μg/m^3)	18.2~46.5	37.7	27.2	145~310	178~280	100~270

Source: Dockery et al.1993; Pope et al.1995; Pope et al.2002; Jing et al.1999; Xu et al.1996; Wang et al.2003.
Note: Beta is the percentage increase in health effect per 1μg/m3 increment of PM_{10}.

Limitation of study areas

The meta-analysis by Aunan and Pan (2004) was primarily based on the results from several large cities and not on middle and small cities. However, the characteristics of air pollution in middle and small cities are often different from those in large ones, and the age structure and susceptibility to air pollution of the local population may also vary with city size. So the extrapolation of the exposure-response functions to the other cities should be considered carefully.

Limitations of methodology

Large-sample epidemiological cohort studies, similar to those carried out in the U.S. to study the effects of long-term exposure to PM on mortality, have not been undertaken in China. Most cross-sectional studies in China have been ecological studies, in which no detailed information at the individual level is collected. This implies that the studies in different locations may not be comparable due to site-specific characteristics. More importantly, the studies do not control for confounding factors that may affect mortality (such as socioeconomic status), which may also be correlated with air pollution. For this reason, we rely on the results of cohort studies from the U.S. (1995, 2002) in the manner described below.

Table 2.8 Exposure-Response Relationships of PM_{10} and Morbidity Outcomes

Health Endpoints	Diseases	Beta	Standard Errors
Hospital admission	RD	0.12	0.02
	CVD	0.07	0.02
New Cases	Chronic Bronchitis	0.48	0.04

Source: Aunan and Pan (2004).
Note: Beta is the increased percent of health effects per μg/m3 increment of PM_{10}.

Proposed Exposure-Response Coefficients

Exposure-response coefficients for long-term exposure and mortality

Since impacts on all-cause mortality are reported both in long-term cohort studies and ecological studies (the latter presumably representing the chronic effects of air pollution on mortality rates), we select all-cause mortality as an endpoint in our assessment.

There are indications that the percentage change in the mortality rate per 1μg/m^3 increment of PM_{10} changes with the concentration level. The studies in the U.S. are all carried out in areas with lower PM_{10} concentrations compared to Chinese studies. The relative risk (*RR*) of dying at a PM_{10} concentration of *C*, compared to the threshold concentration of 15 μg/m³ for the studies reported in Table 2.6, is given by equation (2.1)

$$RR = \exp(\beta C)/\exp(\beta 15) = \exp(\beta \Delta C) \qquad (2.1)$$

where ΔC is the difference between the current PM_{10} concentration and the threshold. This function is plotted in blue in Figure 2.2 for β = .0024 from the Pope et al. (2002) study. The Pope relative risk function reaches 1.38 at a concentration of 150 μg/m3, implying (as explained below) that 28 percent of deaths are premature deaths attributable to air pollution. This is clearly an implausible result. WHO (2004) dealt with this issue by assuming that the RR function becomes horizontal at approximately 100 μg/m3 of PM_{10}, as shown in pink on the graph. This assumption implies that there are no health benefits from reducing PM_{10} from 150 to 100 μg/m3!

One alternative is to use as the RR function equation (2.1) with β = .0012 from a meta-analysis of cross-sectional Chinese studies (see figure 2.3.) These studies, however, were conducted in cities where average PM_{10} levels were well above 150 μg/m3 and may not be applicable to PM_{10} levels below 150 μg/m3. A compromise solution is to assume that exposure is linear in the logarithm of PM_{10} (Ostro 2004), implying that the relative risk function is given by

$$RR = \exp(\alpha + \beta \ln C)/\exp(\alpha + \beta \ln 15)$$
$$= (C/15)^{\beta} \qquad (2.2)$$

Ostro adds 1 to both concentrations, to avoid taking the logarithm of zero, so that equation (2.2) becomes:

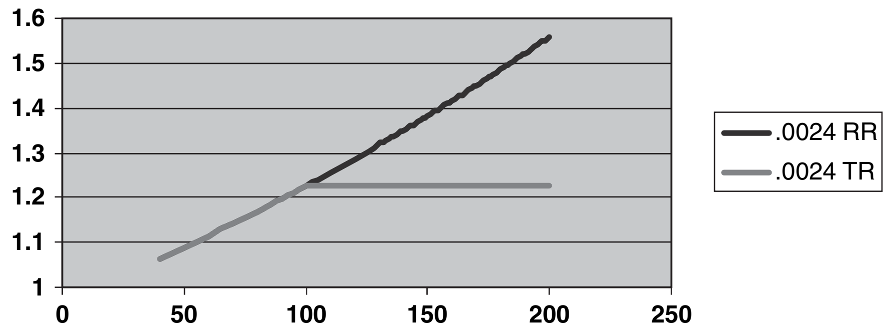
$$RR = ((C+1)/16)^{\beta} \qquad (2.3)$$

When the data from Pope et al. (1995) are fit to the log linear relative risk function, β = 0.073 (s.e. = 0.028) (personal communication from Rick Burnett, July 2006). This relative risk function (labeled Ostro RR) is plotted in Figure 2.3. It coincides with the RR function based on (2.1) with β = .0012 at 150 μg/m3 and yields higher relative risks at lower PM_{10} levels. Figure 2.3 compares this RR function with the RR function implied by equation (2.1) with β = .0012. Equation (2.3) is used to compute the relative risks of PM_{10} concentrations in the CECM.

Exposure-response coefficients for hospital admissions

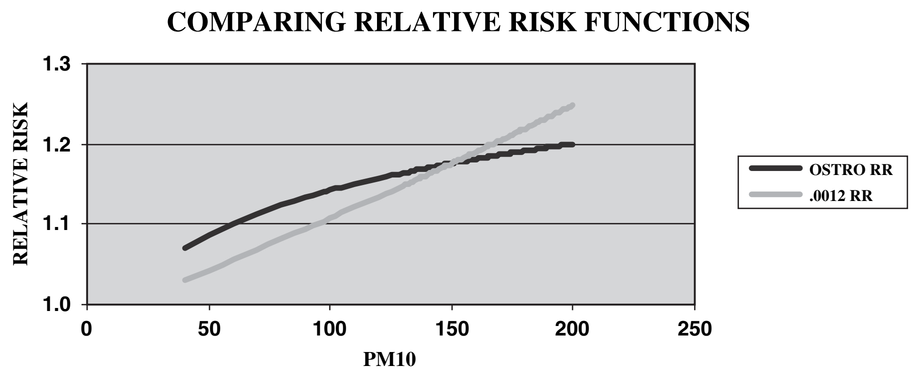
Few studies have been carried out in China addressing hospitalization associated with air pollution (HEI 2004; Aunan and Pan 2004). We

FIGURE 2.2 Comparing Relative Risk Functions



Source: Authors calculation.

FIGURE 2.3 Relative Risk Functions Based on U.S. and Chinese Studies



Source: Authors calculation.

apply the functions derived in Aunan and Pan (2004) to estimate the number of annual excess cases of hospital admissions for cardiovascular diseases and respiratory diseases. The functions are based on two time-series studies in Hong Kong and indicate a 0.07 percent (S.E. 0.02) increase in hospital admissions due to cardiovascular diseases per μg/m^3 PM_{10} and a 0.12 percent (S.E. 0.02) increase in hospital admissions due to respiratory diseases per μg/m^3 PM_{10}. The relative risks for hospital admissions are given by (2.1) with the values of $\beta = .0007$ and $\beta = .0012$, respectively.

Exposure-response coefficients for chronic bronchitis

Aunan and Pan (2004) report an exposure-response coefficient of 0.48 percent (S.E. 0.04) per μg/m^3 PM_{10} for bronchitis in adults and 0.34 percent per μg/m^3 PM_{10} (S.E. 0.03) for bronchitis in children. Altogether, eight cross-sectional questionnaire surveys addressing a range of persistent/chronic respiratory symptoms and diseases were included in Aunan and Pan (2004). All surveys were carried out in Chinese cities, and covered both urban and suburban areas. The coefficients for bronchitis are the result of a meta-analysis of the sub-sample of odds ratios estimated for this particular endpoint (given for Lanzhou, Wuhan, and Benxi). In the studies, the definition of bronchitis was not precise in terms of ICD-9 (or ICD-10) code, but was described as “chronic” or “diagnosed by a physician.” We assume that the endpoint approximates chronic bronchitis, and use the relative risk function (2.1) with $\beta = .0048$ for chronic bronchitis in adults.

CALCULATING HEALTH DAMAGES

With the identified health endpoints and exposure-response coefficients proposed earlier, the health effects from PM_{10} pollution consist of three parts: (1) all-cause premature death; (2) hospital admissions for respiratory disease (RD) and cardiovascular disease (CVD); and (3) new cases of chronic bronchitis.

The number of cases of each health endpoint attributed to air pollution (E) is calculated as the size of the exposed population (P_e) times the difference between the current incidence rate (f_p) and the incidence rate in a clean environment (f_t) [equation (2.4)]. The latter is calculated from the fact that the current incidence rate equals the “clean” incidence rate times the relative risk, *RR*. Substituting (2.6) in (2.5) implies that excess deaths are the product of current deaths (f_pP_e) times the fraction of deaths attributable to air pollution—(*RR-1*)/*RR*. Formally,

$$E = \left(f_p - f_t\right)P_e \tag{2.4}$$

$$f_p = f_t * RR \tag{2.5}$$

implying

$$E = \left((RR - 1)/RR\right) f_p P_e \tag{2.6}$$

Calculation of Baseline Incidence (f_p)

Hospital admissions

The *Health Statistical Yearbook* (Ministry of Health 2004) provides only all-cause hospital admissions by province, and not hospital admissions for specific diseases such as respiratory disease. Another problem is that hospital admissions by province include both rural and urban areas, whereas only the urban population is used to calculate the health costs of air pollution. We estimate hospital admissions for respiratory disease in urban areas in two steps. First, we estimate the number of hospital admissions due to respiratory diseases by multiplying all-cause hospital admissions by the ratio of respiratory diseases to all diseases by province. The percentage of respiratory disease to all diseases is reported in the Analysis Report of the *Third National Health Services Survey* (Ministry of Health Statistical Information Center 2003). This is based on an assumption that the share of patients being admitted to the hospital for respiratory diseases resembles the share of people suffering from respiratory diseases among all people who are ill. Second, we estimate the number of hospital admissions due to respiratory disease in the urban population from the ratio of the urban population to the total population. This is based on an assumption that the hospitalization rate per case of disease is the same in urban and rural areas, which is a crude approximation. Annex A at the end of this chapter presents a detailed description of the data sources.

Premature mortality

Current mortality rates, which vary by city size, are obtained from the *China Health Statistical Yearbook*.

Chronic bronchitis

Calculating annual cases of chronic bronchitis associated with air pollution requires an estimate of the incidence of chronic bronchitis by city. Because only prevalence rates are available, we approximate the annual incidence of chronic bronchitis by dividing the prevalence rate by the average duration of the illness (23 years). This yields an incidence rate of approximately 0.00148.

Excess Cases of Premature Mortality and Morbidity Attributable to Air Pollution

By combining baseline cases of each health endpoint with the selected relative risk functions, we arrive at estimates of the number of excess cases of premature mortality, hospital admissions, and chronic bronchitis attributable to PM_{10}. In addition to calculating the mean number of cases attributable to outdoor air pollution, the 5th and 95th percentiles of cases are also calculated. The monetary value of these damages is presented in Chapter 4.

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3

Health Impacts of Water Pollution

The poor quality of China's scarce water resources, which is increasingly attributed to nonpoint sources such as agricultural runoff and municipal wastewater, has a significant health impact. The impact is particularly high in rural areas, where about 300 million people lack access to piped water, as well as among vulnerable groups, such as children under 5 years of age and women. This study attributes excess cases of diarrhea and excess deaths due to diarrhea among children under 5 in rural areas (only in rural areas) to lack of safe water supply. The study also estimates the number of cancer deaths in rural areas that are due to the use of poor quality surface water as a drinking water source.

China is facing a severe water shortage. In 45 percent of the national territory, annual precipitation is less than 400mm (Zonggu Zhang and Dehong Shi). With a rapidly growing economy and burgeoning populations, the country's scarce water resources are seriously affected by pollution from the vast discharges of industrial and domestic wastewater, indiscriminate solid waste disposal, and runoff from an agricultural sector characterized by excessive use of fertilizer and pesticides and large-scale livestock breeding. Some 300–500 million people in rural areas do not have access to piped water and are exposed to severe health risks related to polluted drinking water. Most urban residents have access to piped water that has been subject to treatment. In smaller cities and townships, the drinking water quality guidelines are frequently violated in piped water and—to an even larger extent—in nonpiped types of water.

The complex and fragmented system for monitoring drinking water resources (using different classification systems and sometimes showing contradictory patterns) complicates a comprehensive assessment of the health effects of polluted drinking water. In this chapter, we have attempted to quantify the health burden related to water pollution on excess diarrhea morbidity and mortality in children under 5 years of age as well as water-related cancer mortality in the general population. Although it seems clear that there are large health risks associated with water pollution in China, it could well be that the lion's share of the costs to society of polluted drinking water are avoidance costs—ranging from the cost of building treatment plants to the cost to households of buying bottled water and small-scale treatment devices.

Water quality is monitored in more than 2,000 river sections across the main rivers in China (MWR 2005). About 25,000 km of Chinese rivers failed to meet the water quality standards for aquatic life and about 90 percent of the sections of rivers around urban areas were seriously polluted (MWR 2005).

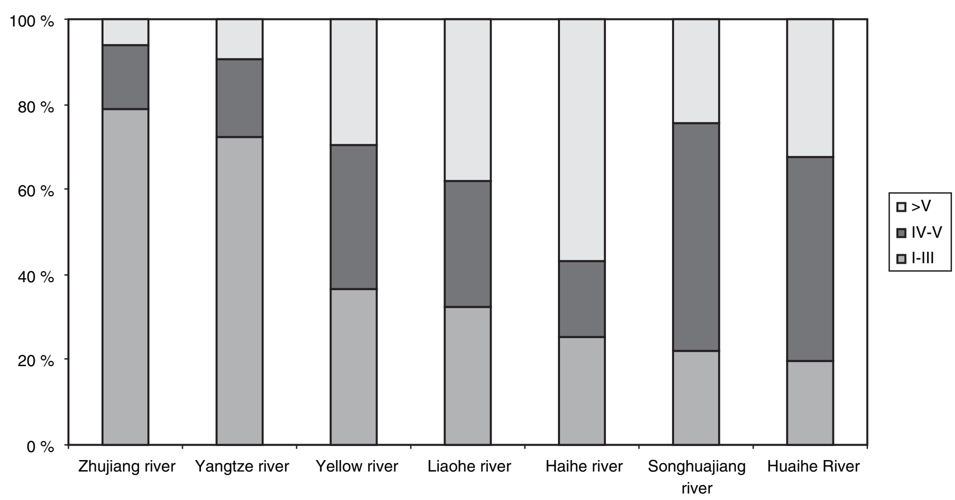
Many of the most polluted rivers have been void of fish for many years. Among the 412 sections of the seven major rivers monitored in 2004, 42 percent met the Grade I–III surface water quality standard, 30 percent met Grade IV–V, and 28 percent failed to meet Grade V (see figure 3.1).[1] Among these seven rivers, the Zhujiang River (79 percent) and Yangtze River (72 percent) had the largest share of sections meeting Grade I–III. The Haihe River was the most polluted, with 57 percent of the monitored sections failing to meet Grade V (Figure 3.1). Major pollutants contributing to poor water quality are ammonia nitrogen, oxygen-demanding organic substances (measured by BOD_5), permanganate value, and toxic petroleum compounds (SEPA 2004). Water pollution has penetrated beyond infecting the surface water found in lakes, rivers, and streams, and over half of the cities now have polluted groundwater (Siciliano 2005).

The amount of wastewater discharged from larger, regulated industries has leveled off since the early 1990s due to an increase in the number and capacity of industrial wastewater treatment facilities. However, discharges from the numerous town and village enterprises (TVEs) and municipal sources are increasing rapidly and are causing extensive pollution of water bodies.

The main pollutants are changing from heavy metals and toxic organic chemicals, which are typically related to discharges of industrial wastewater, to pollutants from nonpoint sources. Runoff from agriculture, including pesticides and fertilizers, is the single greatest contributor to nonpoint-source pollutants. The consumption of chemical fertilizers nearly doubled in the period 1990–2004, and in the same period the use of nitrogenous fertilizers grew by 40 percent (China Statistical Yearbook 2005). Nonpoint sources are difficult to monitor and in many cases more difficult to control than point sources (Yu et al. 2003).

Treatment of domestic sewage has been limited until recently but is increasing steadily. In 1999, China had 266 modern wastewater treatment plants with treatment capacity accounting for only 15 percent of the total 20.4 billion

FIGURE 3.1 Water Quality in Seven Major Rivers in China (percentage of river sections in different water quality classes)



Source: SEPA, 2004

tons of domestic sewage. By the end of the tenth Five-Year Plan period (2001–2005), the government put special emphasis on improving the situation in the three-river, three-lake drainage area. ('The three rivers' are Huaihe, Liaohe and Haihe, and 'the three lakes' are Taihu, Dianchi and Chaohu. The total drainage area of the three rivers and three lakes is 810,000 sq km, traversing 14 provinces with a total population of 360 million.) Four-hundred-and-sixteen sewage treatment plants in these areas have been completed or are under construction, with a daily treatment capacity of 20.93 million tons (State Council 2006). By the end of 2004, the rate of urban sewage treatment in China had reached 46 percent.

As evident from Figure 3.2, the major river systems in the North are more heavily polluted than those in the South, due to serious water scarcity in northern China (MWR 2005). Pollution levels are particularly high in the case of ammonia nitrogen, dissolved oxygen, BOD and permanganate, with 17–33 percent of monitoring sites not meeting class III drinking water quality, mainly in the North (see figures 3.2A–2H). Moreover, population densities are higher in the North, thus implying larger discharges of municipal wastewater into the rivers (see figure 3.3).

Among the 27 major lakes and reservoirs being monitored in 2004, none of them met the Grade I water quality standard; only two met the Grade II water quality standard (7.5 percent); five met the Grade III quality standard (18.5 percent); four met the Grade IV quality standard (14.8 percent); six met the Grade V quality standard (22.2 percent), and ten failed to meet the Grade V quality standard (37.0 percent). The "Three Lakes" (Taihu Lake, Chaohu Lake, and Dianchi Lake) were among the lakes failing to meet the Grade V water quality standard. The main pollution indicators contributing to poor water quality were total nitrogen and total phosphorus (SEPA 2004).

DRINKING WATER—ACCESS AND QUALITY

There is a close relationship between water and health. Water is an essential ingredient for maintaining human life, but, when contaminated, it is also an important medium for the spread of disease. To what extent polluted water resources actually have an impact on people's health depends on the society's capacity to treat sewage and industrial discharges and to purify drinking water. It also depends upon whether people take action on an individual level to avoid consuming polluted drinking water when treatment is absent or not satisfactory. most people in China's urban areas have access to piped water—95 percent in 2003, according to the *China National Health Survey* (CNHS).[2] The corresponding figure in rural areas is unclear. According to the survey, 34 percent of the rural population has piped water, while the reported average percentage on a national level is about 50 percent (Table 3.1). These estimates are close to those based on data from the nationwide *China County Population Census* in 2000, which reported that 33 percent of rural households had access to piped water and 54 percent of the population in China on average had access to piped water.

The census data, which is based on a survey of about 33 million households across rural and urban China, also indicate that a large share of the population does not have access to adequate sanitary facilities, another factor of importance for the spread of waterborne infectious diseases (Table 3.2) (ACMR 2004). The health survey, supported by the census data, implies that about 500 million people in rural areas do not have access to piped water. According to the Ministry of Health, however, 61 percent of the rural population had piped water in 2004 (Ministry of Health 2005), which implies that about 300 million people in rural areas did not have access to piped water. This is in accordance with figures from the Ministry of Water Resources.[3] Whatever the real figure is, it is clear that a substantial number of people in rural areas still rely on well

FIGURE 3.2 Overall Water Quality 2004

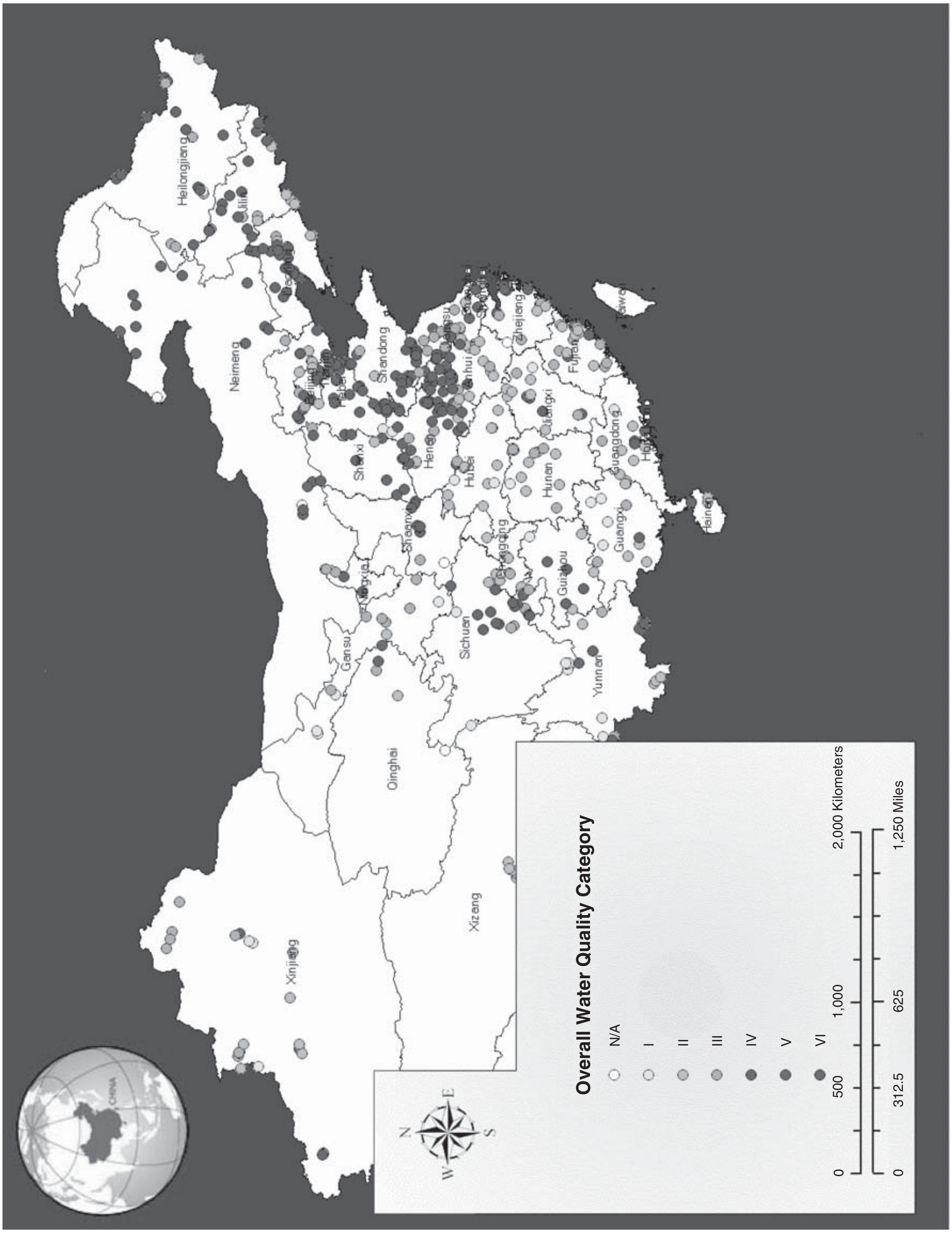


FIGURE 3.2 Overall Water Quality 2004 (*continued*)

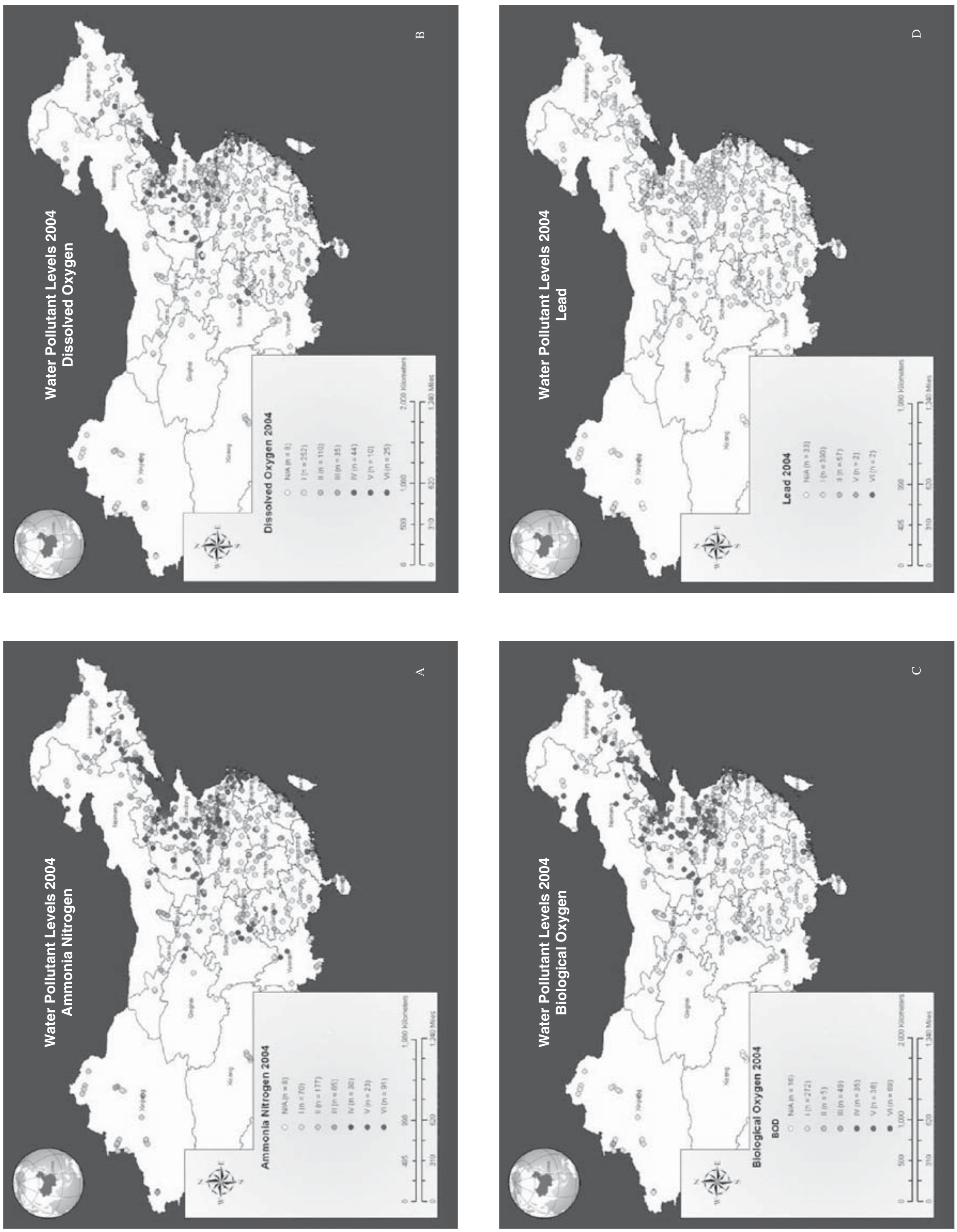


FIGURE 3.2 Overall Water Quality 2004 (*continued*)

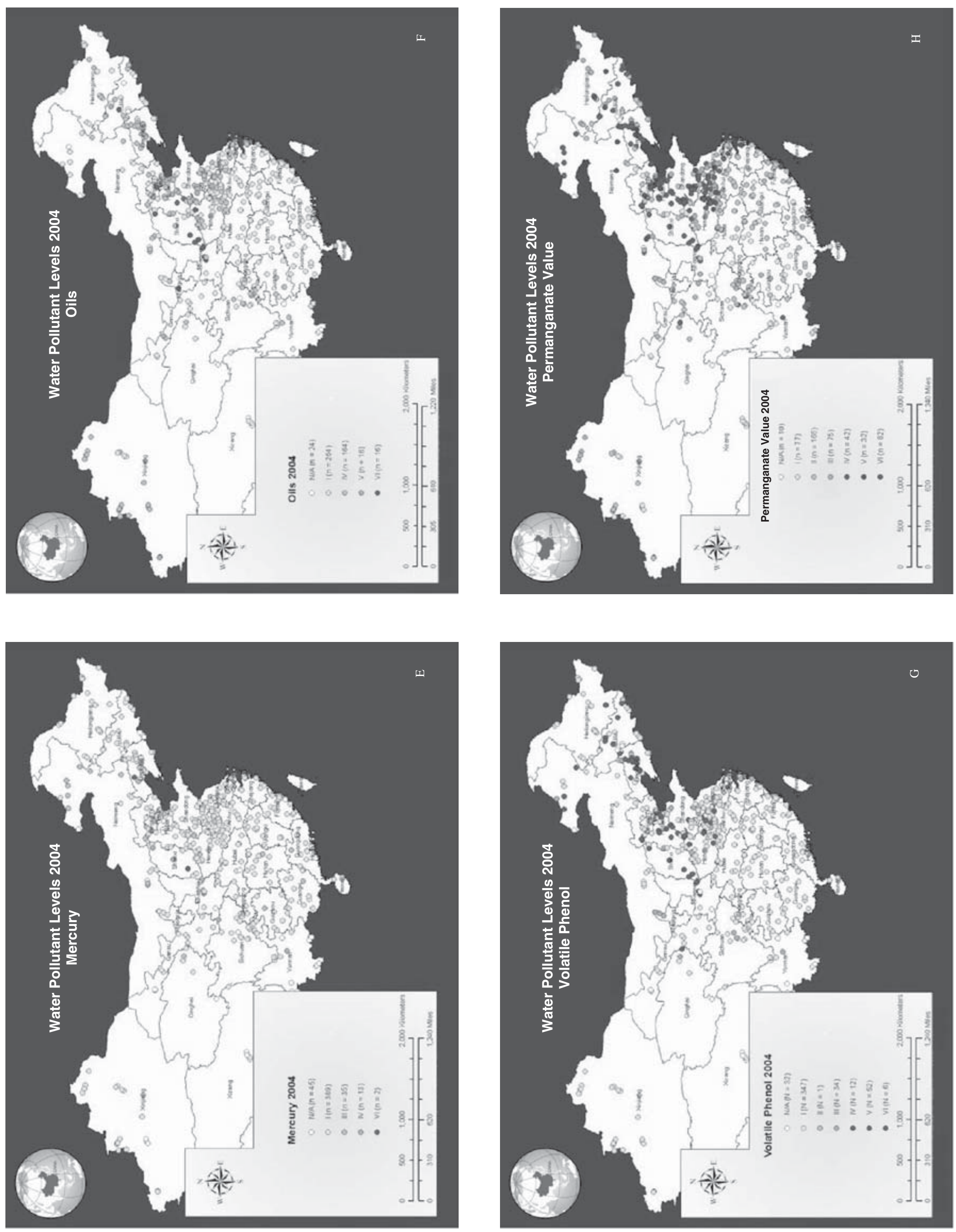
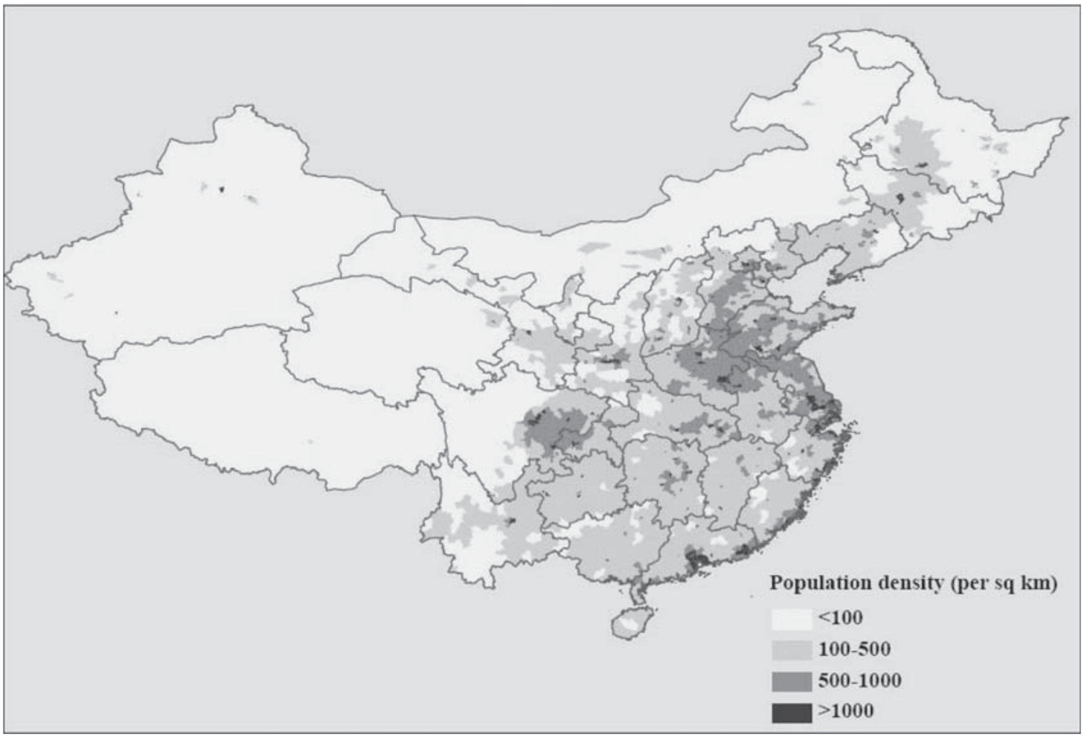


FIGURE 3.3 Population Densities in China (number of people per square kilometer)



Source: China County Population Census (ACMR 2004).

water or water from rivers, lakes, and ponds. Figure 3.4 shows the share of households with access to piped water for each county in China (census data).

Having access to piped water, however, does not guarantee access to clean drinking water. Likewise, dependence on other drinking water types may not necessarily imply a health risk if the water source is protected from contamination. The coverage and technologies of treatment facilities for piped water differ significantly across regions in China. The most comprehensive treatment entails

BOX 3.1 Pollution Accidents

In addition to the continuous discharges of pollution into river systems, accidents may lead to temporary high levels of pollutants. In the aftermath of the accident in the Songhua River in northeastern China in November 2005, where a chemical plant released about 100 tons of the highly toxic chemical benzene into the river, the government carried out an inspection of 127 major chemical and petrochemical plants. The inspection found that many plants were located too close to major bodies of water and that 20 of the inspected plants had serious environmental safety problems. These plants included oil refineries and ethylene and methanol factories along the Yangtze River, the Yellow River, and the Daya Bay near Hong Kong. In the period November 2005 to April 2006, 76 more water pollution accidents were reported by the Chinese government (Associated Press 2006).

TABLE 3.1 Proportion of Drinking Water Types Among Households (urban and rural)

Water Source	Percent of Households
Piped water	49.7
Hand pump	25.8
Well	6.5
Rain collection	2.6
Other (surface)	15.4
Total	100.0

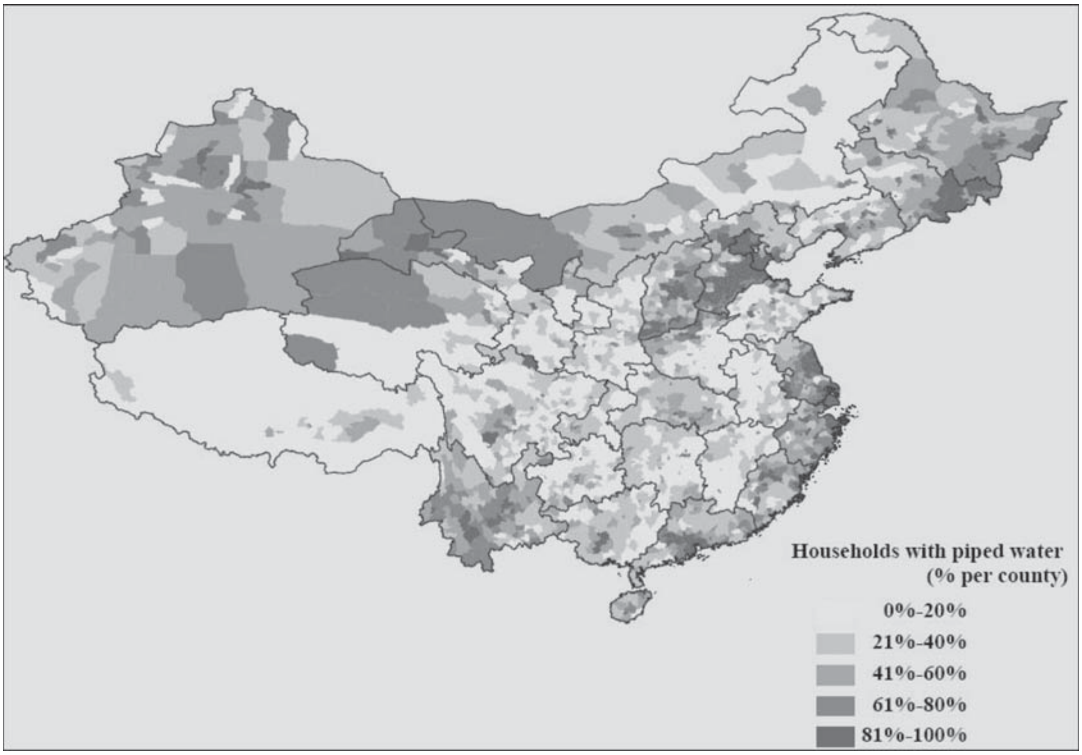
Source: 3rd National Health Service Survey 2003.

TABLE 3.2 Availability of Tap Water and Sanitary Facilities in China (% of total households surveyed)

Water type	
Piped water	45.7
Nonpiped water	54.3
Bathing facility (hot water)	
Centralized support of hot water	0.9
Water heater in home	15.4
Other facility for bath	9.7
No facility for bath	74.0
Lavatories	
WC in home	18.0
Sharing WC with neighbors	0.7
Other type of lavatory in home	49.3
Sharing other type of lavatory with neighbors	3.9
No lavatory	28.0

Source: China County Population Census (ACMR, 2004).

FIGURE 3.4 Households with Access to Piped Water (% of total in county)



Source: China County Population Census (ACMR 2004).

both sedimentation and disinfection, while much of the piped water is subject only to partial treatment (either sedimentation or disinfection). The simplest form of treatment is chlorination.

As seen in Figure 3.5, the level of coliform bacteria is lower in groundwater than in most other drinking water sources. The level of fluoride and arsenic, however, is highest in groundwater. With respect to the mean values for all 300 counties for the different drinking water types, drinking water quality guidelines (Class I) are violated only for total bacteria and total coliform bacteria (see table 4).

The violations are large for some water types. Among the other indicators, however, there is large variability in the measurements for many water types, implying that guidelines must be violated in many samples (see figure 3.5). It should be noted that Class I of the drinking water quality guidelines (Table 3.3) represents the national standard and applies to urban areas. This means that piped water produced by treatment plants in urban areas should not violate the Class I standard. In rural areas, Class III applies.

As evident from Figure 3.5, nitrate is leaking into groundwater, since the level in groundwater and spring water is not lower than in surface water bodies.

CAUSAL AGENTS AND IMPACT PATHWAYS

Water pollutants can be categorized into two main types—*biological pollutants* (including microorganisms causing infectious hepatitis A or E, dysentery, typhoid fever, cholera and diarrhea),[4] and *chemical pollutants* (including inorganic substances such as nitrates, phosphates, mercury, arsenic, chrome, fluorine and lead, and organic compounds such as phenols, benzene and other aromatic compounds, and oil). While infectious diseases typically occur as an acute effect, exposure to biological pollutants may also have long-term health implications. For instance, chronic intestinal infections may develop and infectious hepatitis may in the long run lead to cirrhosis and liver cancer. High levels of chemical pollutants can cause acute poisoning, whereas—perhaps more importantly—long-term exposure to lower levels may lead to chronic health effects such as cancers and may enhance the risk of adverse pregnancy outcomes such as spontaneous abortions and birth defects.

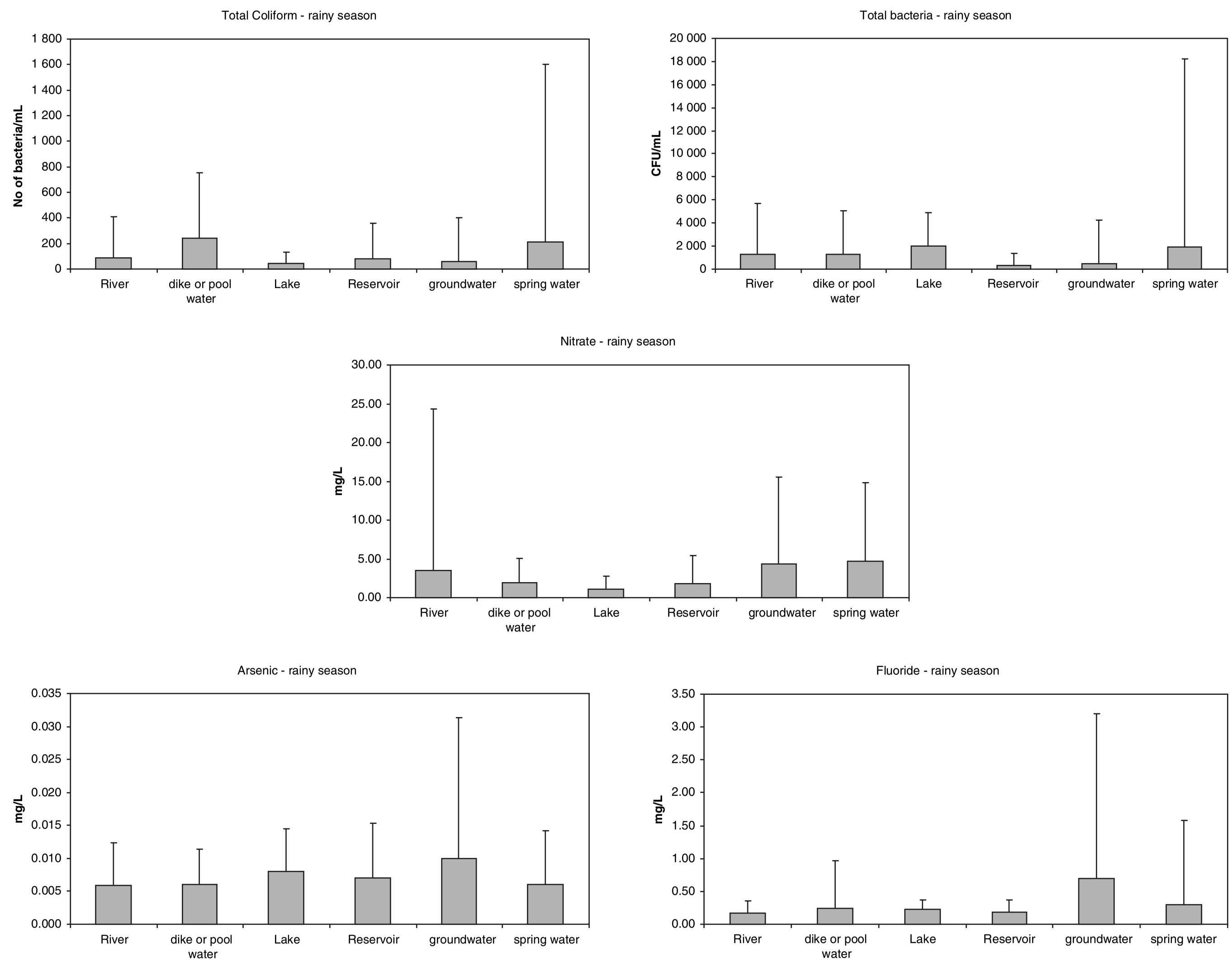
The disease matrix presented in Figure 3.6 on the next page summarizes the different health outcomes that may result from drinking polluted water. The matrix is based on information collated from WHO's *Guidelines for Drinking Water Quality* (WHO 1996) for a number of biological and chemical pollutants, as well as a comprehensive literature search, which was conducted in order to investigate associations between biological/heavy metal pollutants in drinking water and possible health outcomes. Most of the retrieved studies gave a measure of effect or provided the means to calculate it (see appendix 1 for details of the studies). The objective of this exercise was not to retrieve all possible publications on the association between drinking water pollutants and health outcomes (i.e. this was not a systematic review), but rather to collect enough evidence

TABLE 3.3 Drinking Water Quality Standards for China

	Class I	Class II	Class III
Chrome (degree)	15.0	20.0	30.0
Turbidity (degree)	3.0	10.0	20.0
Total dissolved solids (mg/L, CaCO3)	450.0	550.0	700.0
Iron (mg/L)	0.3	0.5	1.0
Manganese (mg/L)	0.1	0.3	0.5
COD (mg/L)	3.0	6.0	6.0
Chlorate (mg/L)	250.0	300.0	450.0
Sulfate (mg/L)	250.0	300.0	400.0
Fluoride (mg/L)	1.0	1.2	1.5
Arsenic (mg/L)	0.1	0.1	0.1
Nitrate (mg/L)	20.0	20.0	20.0
Total bacteria (/mL)	100.0	200.0	500.0
Total coliform (/L)	3.0	11.0	27.0

Source: *SEPA*

FIGURE 3.5 Water Quality in the Rainy Season (April–October) in Different Sources of Drinking Water in Rural Areas, 2004 (total coliform, total bacteria, nitrate, and arsenic)



Source: CDC, Beijing.

Note: *These data should be interpreted with caution as monitoring of rural drinking water is relatively new and there may be errors in the data.*

to show possible associations. Studies relating to mortality as an outcome have not been incorporated in the presented matrix.

From the matrix, it is clear that there are positive associations between exposure to chemical pollutants, namely nitrate/nitrite and arsenic, and at least nine different malignancies. However, studies on arsenic seemed to be more established than those on nitrate/nitrites. Arsenic is responsible for inducing several malignant tumors affecting epithelial tissues including skin, liver, lung, bladder and kidney. It is also associated with cardiovascular, respiratory and neurological disorders.

Much research has been conducted on the effect of nitrate in drinking water on human health; but, with considerable controversy over some of the outcomes investigated, particularly gastric cancers. Nevertheless, several epidemiological studies have demonstrated increased risk for bladder, ovarian and colorectal cancers associated with ingestion of nitrates in drinking water. Furthermore, there is strong evidence that nitrates are also associated with an increased risk of non-Hodgkin's lymphoma.

Many epidemiological studies have looked into the impact of prolonged ingestion of fluoride in drinking water and have concluded that it primarily affects skeletal tissues in different degrees depending on the concentration levels detected. At concentrations as low as 0.9–1.2 mg/liter, fluoride could cause dental fluorosis. However, with higher concentrations, it may cause skeletal fluorosis and with even more elevated levels it may result in crippling skeletal fluorosis. There is inconclusive evidence on fluoride carcinogenicity in humans.

Lead has been shown to cause renal disease and may in some cases be associated with chronic nephropathy, particularly with prolonged exposure. Furthermore, strong associations have been documented between blood lead levels in the range of 7–34 microgram/dl and hypertension. From another perspective, there is very little evidence, if any, that lead is a human carcinogen.

The international literature clearly documents associations between fecal coliforms/total bacteria and diarrhea. Moreover, associations between these biological pollutants and other digestive outcomes such as cholera, dysentery, gastroenteritis, giardia, salmonella, typhoid, and shigella have also been established

HEALTH AND CHEMICAL WATER POLLUTANTS

Chemical pollution of water resources may be due to natural conditions. In China, chronic endemic arsenism is among the most serious endemic diseases related to drinking water (Xia

TABLE 3.4 Exceeding Drinking Water Quality Standards for Total Bacteria and Total Coliform Bacteria in Drinking Water Types in China (ratio between the mean value of samples from 300 rural counties and the guideline value, Class I)

Water Quality Indicator	Piped Water (treated)	Piped Water (partially treated)	Piped Water (untreated)	Nonpiped (by machine)	Nonpiped (manual)	Nonpiped (hand pump)
Total bacteria	6.6	7.2	5.8	8.3	8.0	4.9
Total coliform	11.4	23.7	18.7	20.4	103.1	12.4

Source: National CDC, Beijing.
Note: *These data should be interpreted with caution as monitoring of rural drinking water is relatively new and there may be errors in the data.*

FIGURE 3.6 Matrix for Biological/Chemical Drinking Water Pollutants and Health Outcomes

Health Outcome	Biological Pollutants		Chemical Pollutants			
	Fecal coliform	Total bacteria	Nitrate/ Nitrite	Fluoride	Lead	Arsenic
Malignancies						
Bladder Cancer			•			•
Colorectral Cancer			•			
Gastric Cancer			•			
Liver Cancer						•
Lung Cancer						•
Renal Cancer						•
Skin Cancer/ Pre-malignant Lesions						•
Ovarian Cancer			•			
Non-Hodgkin Lymphoma			•			
Cardiovascular						
Peripheral Vascular Disease						•
Hypertension		•			•	•
Respiratory						
Bronchiectasis						•
Bone/Skeletal Deformities						
Bone Deformity				•		
Dental Fluorosis				•		
Skeletal Fluorosis				•		
Neurological						
Central Nervous System Defects					•	
Mental Retardation					•	
Peripheral Neuropathy						•
Digestive						
Cholera	•					
Diarrheal Diseases	•	•				
Dysentery	•	•				
Hepatitis	•	•				
Typhoid Fever	•	•				
Hepatomegaly (enlarged liver)						•
Pregnancy-Related						
Adverse birth outcomes					•	•
Spontaneous Abortion						•
Other						
Renal Dysfunction		•			•	
Diabetes Mellitus						•

Source: WHO 1996 and authors' calculations.

and Liu 2004). High levels of arsenic (As) in drinking water are attributable to the geological-geochemistry environment. In China, high levels of As in groundwater are mainly found in the (a) plain of the Great Bend of the Yellow River and the Hu-Bao Plain in the Inner Mongolia Autonomous Region; (b) the Datong basin of Shanxi Province; (c) the floodplain of the northern side of the Tian Mountain of Xinjiang Uygur Autonomous Region; and (d) the southwest coastal plain of Taiwan (Lin et al. 2002). Epidemiological studies have shown that high levels of As in drinking water are associated with skin cancers and other cancer, hypertension, and peripheral vascular diseases. Dose-response relationships are reported for some health endpoints. It is estimated that 2.3 million people are exposed to high levels of As (>0.05mg/L)[5] through drinking water (Xia and Liu and references therein, MWR 2005). About 7,500 patients were diagnosed with arsenism in the areas surveyed by the Ministry of Health in 2003 (MoH 2004).

Geological conditions may also lead to high levels of fluorine (F) in drinking water. According to the Ministry of Water Resources (2005), 63 million people drink water with high concentrations of fluorine. Endemic dental fluorosis related to drinking water affected 21 million people in 2003, whereas 1.3 million suffered from drinking-water-related skeletal fluorosis (MoH 2004). Water-pollution-related fluorosis is more prevalent in the northeast and central part of the country, but cases are reported in nearly all provinces.

In the environmental cost model, we do not include the natural water contaminants but focus on anthropogenic pollutants in drinking water and their potential health risks. Although the health effects of natural and anthropogenic pollutants may overlap, and it may be difficult to disentangle the individual contribution of the two types—for instance, for cancers—it is believed that anthropogenic pollution of drinking water is the most important in today's China.

Mortality rates in China for cancers associated with water pollution are shown in Figure 3.7, along with the world average rates. For stomach, liver, and bladder cancer, the rates are highest in rural areas. For liver and stomach cancer in particular, the mortality rates in China are well above the world average. Liver cancer is the most prevalent type of cancer in rural China.[6]

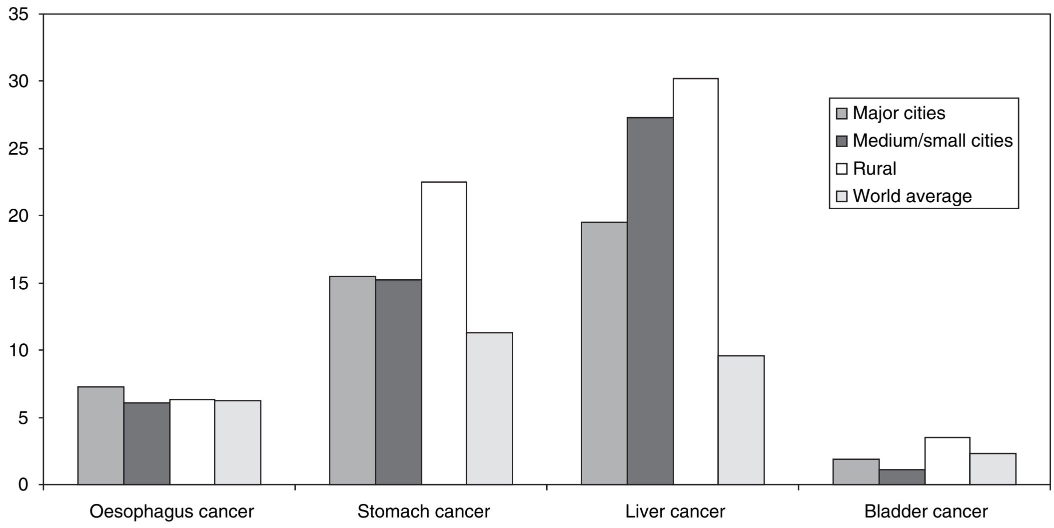
HEALTH AND BIOLOGICAL WATER POLLUTANTS

Drinking contaminated water is typically only one of several ways of contracting infectious diseases. Pathogens may also be spread by food and by flies, and because pathogens are spread by direct contact, hygiene is of primary importance. Figure 3.8 below portrays the famous F-diagram, which captures the potential exposure pathways for fecal-oral transmission.

The F-diagram clearly shows that breaking the fecal-oral transmission route, and thus reducing infections, is not entirely dependent on the availability of clean water but also depends on other factors, including safe disposal of feces (safe sanitation), hygiene behaviors—especially handwashing with soap after defecation—and safe food handling and storage. Generally, disease incidence is high in areas, where basic sanitary facilities are lacking.[7] Water scarcity may also enhance the spread of infectious diseases. Convenient access to sufficient water quantity encourages better hygiene and, therefore, limits the spread of disease.[8]

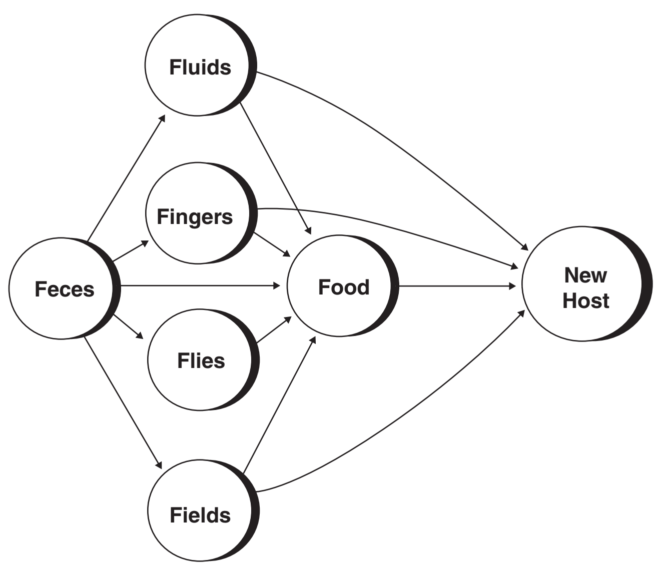
As opposed to the chronic diseases arising from long-term exposures to carcinogenic pollutants, the incidence rates for infectious diseases may vary substantially from one year to the next and from one season to another (*see Box 3.2 for Chongqing study*). Outbreaks of the disease may cause very high incidence rates in an area during a limited period of time. The fatality rates for these diseases are, however, relatively low. The case-fatality rates for dysentery, typhoid/paratyphoid, and cholera in China in 2003 were on average 0.05 percent,

FIGURE 3.7 Mortality Rate for Diseases Associated with Water Pollution (1/100,000) in China, 2003 (world averages in 2000)



Source: MoH, 2004, and WHO, 2006, GLOBOCAN, 2000

FIGURE 3.8 F-Diagram for Fecal-oral Transmission



0.06 percent, and 0.41 percent, respectively. As shown in Box 1, the incidence rate of water-pollution-related infectious diseases is highest in children. The death toll is also highest in children, particularly for diarrheal diseases. In China, the mortality due to diarrhea in children under five in rural areas is nearly twice the rate in urban areas (1.35 vs 0.75 deaths per 100,000 children) (MoH 2004).

Figure 3.9 shows the distribution of cases of Hepatitis A, dysentery, and typhoid/paratyphoid fever across provinces of China in 2003. Generally, higher rates prevail in western parts of the country. Dysentery is the most frequent of the water-related infectious diseases.

In spite of outbreaks every year, the occurrence of dysentery has fallen dramatically in China in the last decades and now seems to be stabilizing (see figure 3.10A). The worst outbreak occurred in 1975, when an incidence rate of 1,000 per 100,000 was reported. As evident from Table 3.5 and Figure 3.10 outbreaks of typhoid/paratyphoid fever are rarer than dysentery, with an inci-

BOX 3.2 Drinking Water and Waterborne Infectious Diseases in Rural Areas in Chongqing

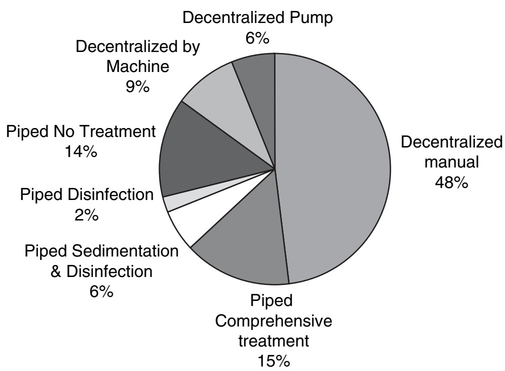
As part of this study, specific work in Chongqing found that the rural population in this province faces many challenges with respect to drinking water supply. Not only do rural areas have limited access to piped water as compared to urban areas, but most of the piped water undergoes little treatment and has significant levels of contamination. This makes the rural population more susceptible to some waterborne infectious diseases.

Drinking Water in Chanqing

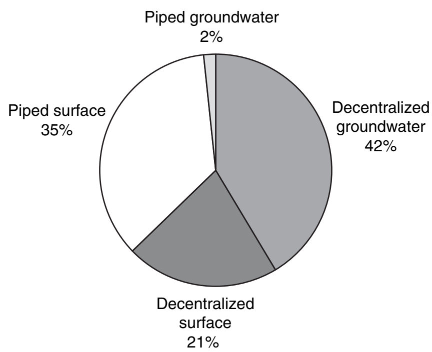
People in Chongqing Province get their main drinking water supply from a variety of sources, including centralized piped water as well as wells, ponds, rivers, and ditches. Generally, the decentralized (nonpiped) water sources undergo no treatment and are less safe for drinking water purposes. Monitoring of drinking water across rural areas in Chongqing shows that the level of the coliform group bacteria (an indicator of fecal contamination) in nonpiped drinking water is about ten times the level in piped water, and there are more frequent incidents of extreme bacteria levels in the rural nonpiped water.

In Chongqing, only around 30 percent of the population has access to piped water (China Census 2000), and, as elsewhere in China, most of this population resides in the largest urban centers. However, only a fraction of the piped water supply undergoes comprehensive treatment before it reaches the end users. Among the 10 counties and urban districts for which detailed information of water supply is available, the share of the population that has access to comprehensively treated piped water is less than a third in all counties/districts, and on average only 15 percent of the population has access to comprehensively treated piped water.

Share of population with centralized and decentralized water system according to a) treatment (for centralized) and means of distribution (for decentralized), and b) source (surface water or underground water)



(a)



(b)

Source: Authors calculations.

The degree of treatment of piped water also varies between urban and rural areas. While drinking water treatment plants in cities and to some extent in smaller townships provide comprehensive treatment of the water through sedimentation and disinfection, a large share of the piped water in townships and villages undergoes only limited treatment (either sedimentation or disinfection or chlorination). The overall effectiveness of treatment is, therefore, very limited. According to 2001–04 monitoring data from 100 township treatment plants in 14 countries upstream of the Three Gorges area, the levels of a number of contaminants—such as arsenic, fluoride, and nitrate—were not significantly affected by treatment. Treatment did reduce the total bacteria content, but the resulting water still had on average 83 percent more coliform bacteria than permitted by the national standard for drinking water quality (Class I). The mercury in the treated water was on average 38 percent above the standard, and the levels of heavy metals like arsenic (As) and cadmium (Cd) were on average approximately 3 percent higher than the standard.

(*continued*)

BOX 3.2 Drinking Water and Waterborne Infectious Diseases in Rural Areas in Chongqing (*Continued*)

The urban/rural discrepancy is also evident in the water sources, which have strong implications on drinking water quality. While the piped water in the 10 counties and urban districts mainly comes from surface water, the decentralized sources are mainly underground water (wells and springs) (see figure above). Most of the underground water is from shallow wells, however, which are easily contaminated by wastewater and runoff from industry, agriculture and households. In figure 1b, the combination of source and treatment most likely associated with the highest risk of infectious diseases is decentralized surface water, on which 21 percent of the population is dependent. In addition, untreated piped surface water probably entails a correspondingly high risk. As nearly 40 percent of the piped water is sent untreated to the end-users, and most of the piped water is from surface water bodies, a substantial share of the total population in these areas—around 13 percent—has untreated surface water in their tap.

The measurements showed that the water quality in Chongqing varies substantially from year to year and between seasons. The study found that the median values of main contaminants, as Coliform bacteria, total bacteria, As, and Hg, did not fluctuate very much from year to year in the period 2001–2004. The mean values did, however, fluctuate considerably, indicating that during some of the years, there were incidents of very high pollution levels for a shorter period of time. For most of the water pollution indicators, the noncompliance rate is higher in the flooding season compared to the dry season.

Waterborne infectious diseases in Chongqing

Hepatitis A, dysentery,[9] and typhoid/paratyphoid fever are three main types of infectious diseases associated with polluted drinking water. Fatality rates for all three diseases in Chongqing are low, indicating that few people die from these diseases, but the annual incidence rates vary. Whereas the incidence rate of Hepatitis A is somewhat higher in Chongqing compared to the average incidence rates in China (12.3 vs. 7.4 cases per 100,000 in 2003), the incidence rates for the two other diseases are lower (for dysentery 27.9 versus 34.5 cases per 100,000, and for typhoid/paratyphoid 0.9 versus 4.2 cases per 100,000) (MoH 2004). The incidence rates are, however, high relative to those found in European countries and the United States, which indicates that there is still a lot of potential for improvement.

The study found that outbreaks of infectious diseases vary considerably from year to year and are generally more frequent in the flooding season as compared to the dry season. As shown in the figure below, there may be large differences between counties when it comes to outbreaks. In Chongqing, outbreaks of the three different diseases occurred independently during the period 2001–04, with no spatial correlation between them.

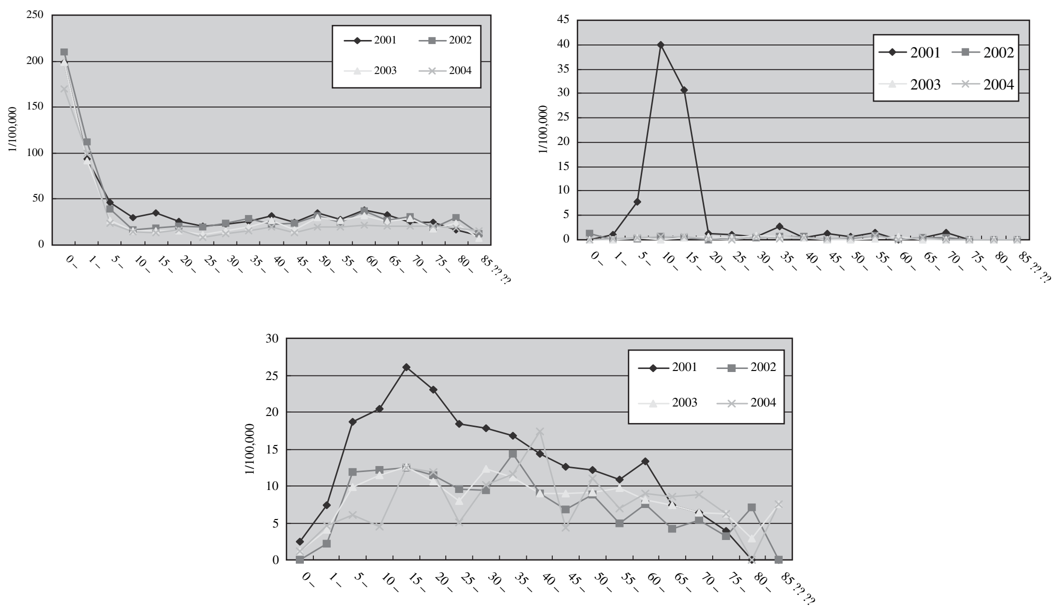
Children in Chongqing are much more likely than adults to contract infections diseases, particularly dysentery and typhoid fever. The incidence rate for children under 5 years of age is 10 percent higher than the rest of the population. For hepatitis A, incidence rates are also markedly higher in children and adolescents; however, a considerable share of the cases also occur in the older age groups (see figure below).

The data available from Chongqing, though limited, show a significant correlation between the level of total bacteria in drinking water[10] and incidence rates for dysentery. While the incidence rate of typhoid was associated with total bacteria to some extent (for females), hepatitis A did not show a clear association with the total level of bacteria or coliform group bacteria as monitored in the rural drinking water.

(*continued*)

BOX 3.2 Drinking Water and Waterborne Infectious Diseases in Rural Areas in Chongqing (*Continued*)

Incidence of (a) Dysentery, (b) Hepatitis A, and (c) Typhoid in 19 Counties in Chongqing



Incidence rates of (a) Dysentery, (b) Hepatitis A, and (c) Typhoid Among Female Age Groups in Chongqing, 2001–04

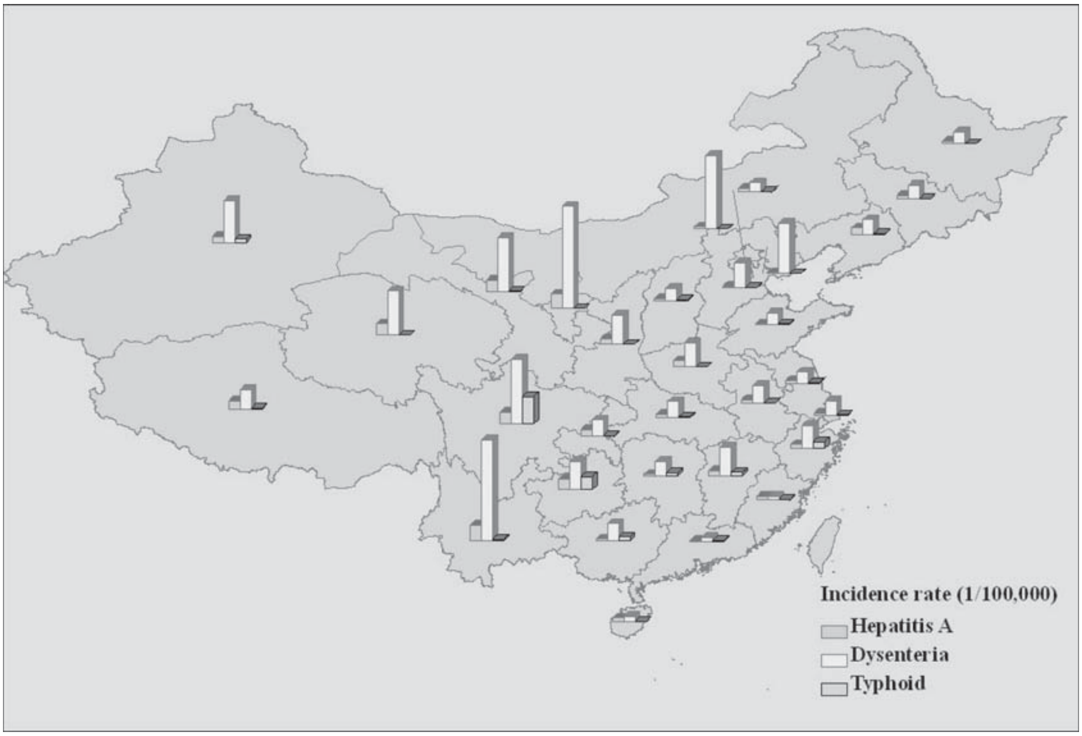
dence rate of around 4 cases per 100,000 in 2003. This rate is also much lower than the world average. The annual incidence of typhoid worldwide at present is estimated to be about 283 cases per 100,000.[11] The incidence rate of cholera has been low in China in recent years—0.02 cases per 100,000 were reported in 2003.

Domestic sewage and agricultural runoff may lead to *eutrophication* of water bodies. In addition to the health risk associated with the eutrophiocation agents themselves, including nitrates and phosphates, eutrophication supports the growth of cyanobacteria that can produce toxins such as microcystins. These are potent liver cancer promoters and are directly hepatotoxic to humans. Microcystins in drinking water cannot be completely removed by common disinfection and heating (MWR 2004 and references therein: Wang et al. 1995; Ling 1999).

CHINESE STUDIES OF HEALTH EFFECTS OF DRINKING WATER POLLUTION

The basis for reliably estimating the full public health implications of drinking water pollution on a population level in China is limited for several reasons. First, there are relatively few studies in China and other developing countries addressing the exposure-response relationship between drinking water pollution and health effects. Water pollution epidemiology and its application is severely hampered by the fact that a range of other factors contribute to disease. Contaminated water is typically one of several ways of contracting infectious disease and is closely linked to sanitation and hygiene, as discussed above. Similarly, in the case of chemical pollutants, contaminated water is also one of several ways of getting ill. Enhanced

FIGURE 3.9 Incidence Rates of Hepatitis A, Dysentery, and Typhoid/Paratyphoid Fever in China in 2003 (1/100,000)



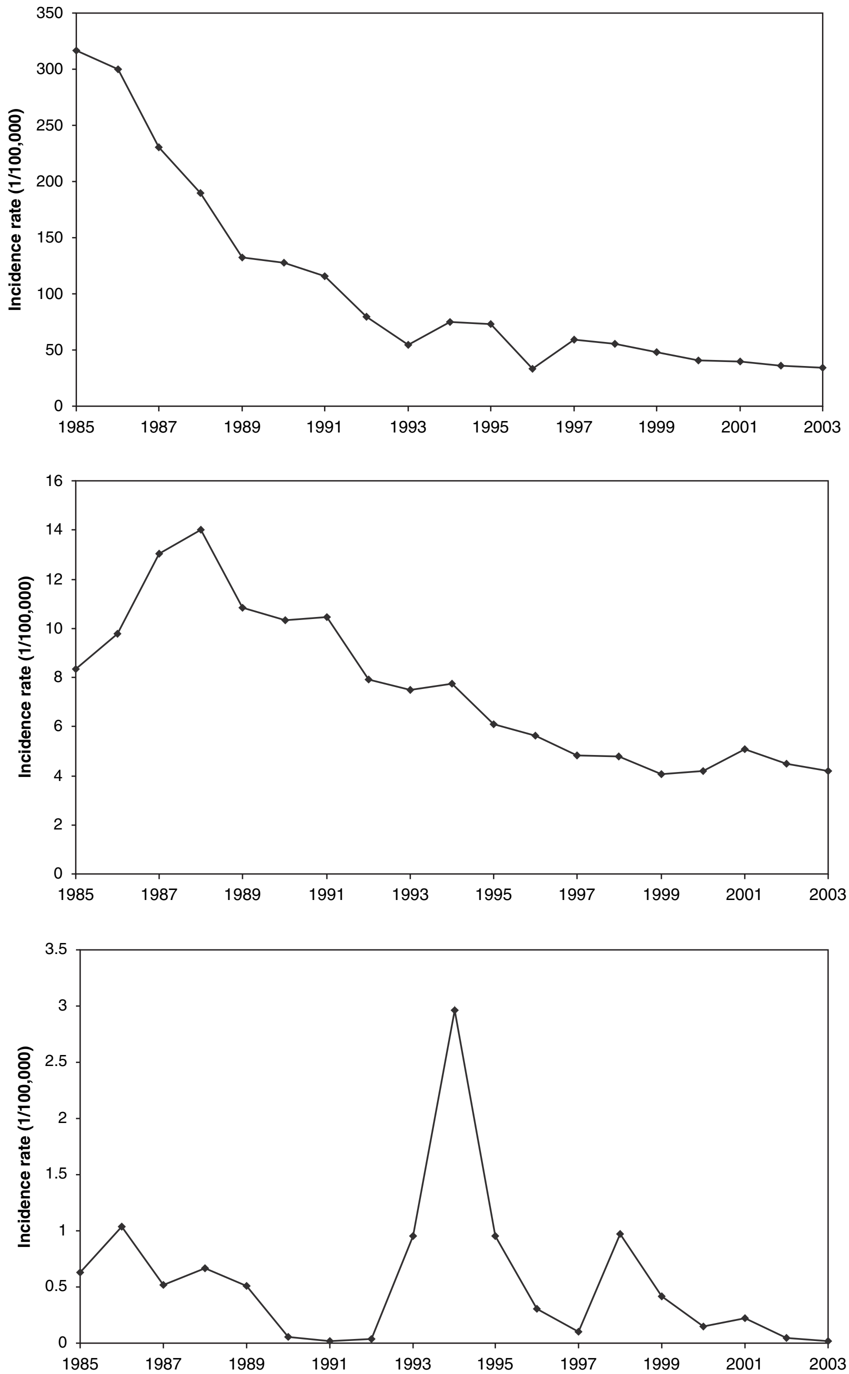
Source: MoH 2004.

rates of disease may as well be related to occupational exposure, smoking, food, and other life-style factors.

Secondly, even when exposure-response functions are available, the exposure assessment is complicated by the fact that—in contrast to air pollution—there is a larger scope for people being protected from exposure. Finally, the assessment of attributable risk from water pollution is complicated by the fact that in many cases the causal agent of disease is not monitored directly, and indicators of exposure are needed. For instance, even though hepatitis A is one of the most prevalent waterborne infectious diseases in developing countries, China included, hepatitis A virus is rarely monitored directly. Instead, fecal bacteria are used as an indicator of possible contamination with hepatitis A virus and other microorganisms. As the environmental resistance, and thus lifetime, of the virus is higher than fecal bacteria, it may not be a quantitative association between the content of virus and bacteria downstream of the discharge area (Fernández-Molina et al. 2004). Clearly, this implies that an exposure assessment based on water quality at a limited number of monitoring sites will be rather uncertain.

In China, most studies addressing health effects from water pollution have looked at drinking water pollution and cancers. Su De-long (1980) explored causal factors of liver cancer in Qidong county of Jiangsu Province, and found that the morbidity of liver cancer was closely related to drinking water contamination. Xu Houquan et al.

FIGURE 3.10 Incidence Rates for Dysentery (A), Typhoid/Paratyphoid Fever (B), and Cholera (C) in China, 1985–2003



Source: MoH 2004.

TABLE 3.5 National Average Incidence Rates of Hepatitis A, Dysentery, Typhoid/ Paratyphoid Fever, and Cholera in China in 2003 (1/100,000)

Disease	Incidence Rate (1/100,000)
Hepatitis A	7.37
Dysentery (viral and amebic)	34.52
Typhoid and paratyphoid	4.17
Cholera 0.02	

Source: MoH 2004.

(1995) carried out a case-control study of risk factors of liver cancer around the Nansi Lake, Shandong Province. They showed that drinking lake water, getting in touch with lake water, drinking alcohol, and eating fish were all risk factors for liver cancer. The estimated odds ratios were 6.55, 3.24, 1.86, and 2.55, respectively. Xu Houquan et al. (1994) carried out a retrospective cohort study on the relationship between water pollution and tumors and showed that the mortality rates of stomach, esophagus, and liver cancer for people drinking lake water were higher than those drinking well water. The relative risks (RR) were 1.56, 1.50, and 1.63. The nationwide study on organic pollution of drinking water and liver cancer by Wang Qian et al. (1992) showed that mortality due to liver cancer for men and women was positively correlated with the chemical oxygen demand (COD) in drinking water. In a 16-year retrospective cohort study in an area with enhanced stomach cancer incidence rates, Wang Zhiqiang et al. (1997) found that mortality due to stomach cancer in people drinking river water was significantly higher than in people drinking well water. Monitoring data showed high levels of ammonia, nitrite, chloride, COD, and heavy metals like lead and mercury, suggesting that drinking polluted water is one of the causal factors of stomach cancer. In Baoding city in Hubei Province, Hu (1994) reported that the mortality rates for liver and esophagus cancer among residents relying on groundwater that was contaminated by sewage was significantly higher compared to people in the control area. Pan and Jiang (2004) investigated the correlation between various water quality indices in drinking water and the mortality rates of a range of cancer types in the Yangtze and Huai He river basins in the period 1992–2000. They found a significant positive correlation between the level of COD (chemical oxygen demand), fluorine, and chloride, and male stomach cancer.

A number of studies have also examined the effects of water pollution on infectious diseases in China. Pan and Jiang (2004) investigated the correlation between coliform group bacteria and the integrated water quality index (IWQI), which includes a wide range of water quality indicators, in drinking water and the incidence rates of infectious diseases in the Yangtze and Huai He river basins in the period 1992–2000. They found significant correlations between the level of coliform group bacteria in drinking water and the incidence rates of diarrhea, and between the IWQI and incidence rates of typhoid/paratyphoid and diarrhea for both men and women. No correlation was found for either index regarding the incidence of hepatitis A. They also show that there was a strong correlation between the level of coliform group bacteria in surface water and drinking water in rural areas in the two river basins. Because monitoring sites were changed during the period, the number of counties for which both disease data and drinking quality data were available was somewhat limited in the study.

In the present study, we found no statistically significant relationship between the level of total coliform bacteria in rural drinking water and incidence rates for infectious diseases. However, due to lack of data, it was not possible to control for the range of possibly important confounding factors in the analysis. The data included in the analysis were incidence rate data for 2004 for infectious diseases from the National Infectious Reporting System (dysentery, hepatitis A, typhoid/

paratyphoid, diarrhea) and data for rural water quality from the National Rural Water Quality Monitoring System provided by the China Center for Disease Control and Prevention (CDC).

VALUATION MODELS IN THE ECM

Estimating Excess Diarrheal Disease Morbidity in Children

Data from the Third National Health Service Survey—prepared by the Health Statistics and Information Center in the Ministry of Health (MoH)—were used to derive exposure-response functions for diarrhea in children under 5 years of age in rural China. The survey was carried out in 95 counties in 31 provinces and incorporated questions pertaining to household characteristics, individual factors, disease prevalence, and costs for treatment. Since no indicators directly related to water quality were included in the survey, we decided to use a conservative approach in the analysis by considering access to piped water as the safest drinking water source. We recognized that piped water does not inherently equal safe, clean water, and that there are sometimes violations of drinking water quality standards, particularly in rural China. But given the absence of data on water quality in the survey, we used piped water as a proxy indicator for safe water. The role of sanitation was considered in the analysis and was controlled for using multivariate modeling. The role of hygiene, however, was not assessed due to the lack of any indicators related to it, especially handwashing. The present analysis and results are based on a rural household-level rather than an individual dataset to mitigate any disease-clustering effects that may exist.

Outcome of interest

Household diarrhea prevalence was estimated by calculating the proportion of households with one or more diarrhea cases in children under 5 years relative to the total number of households with children less than 5 years of age in rural China.

Exposure assessment

The survey categorized drinking water sources into five types, as shown in Table 3.6 below. Although all sources except for surface water (drawn directly from rivers, lakes, pools, canals, ditches, and house drains) are considered to be relatively safe in China, this analysis took a more conservative approach in considering piped water as the only safe drinking water source. An earlier survey estimated that about 50 percent of the population in Class IV rural areas still drinks water not meeting the national sanitary standards.[12] This could mean that at least 30 percent of this population (considering that surface water accounts for nearly 20 percent) are using unsafe water sources that includes hand pumps, wells (i.e. underground water), and rain collection. Therefore, classifying piped water as the only safe source is the most conservative approach.

Sample population description

After data cleaning and reduction, we conducted a descriptive analysis to highlight the socioeconomic and demographic profile of the 7,103 sampled rural households with children less than 5 years of age (see table 3.7).

From the table, it is clear that a small number of the surveyed households (2.9 percent) were officially listed as poor. However, the majority of these households (84 percent) had an income of less than 3,000 RMB, while half of them were spending more than 500 RMB on health-related costs—indicating that a very high share of a limited income was used for health care.

Nearly 88 percent of mothers in these households had some education (most had completed either primary or secondary education). Nearly one-fifth of the surveyed population did not have access to safe sanitation and hence relied on defecation in the open.

Two-week household diarrhea prevalence

The two-week prevalence for household diarrhea in rural China was computed and accounted for 2.2 percent of the total number of households.

TABLE 3.6 Water and Sanitation in Rural China

	Source	Percentage
Water (n=7,036)	Piped Water	33.6
	Hand Pump	33.3
	Well	8.2
	Rain Collection	3.3
	Other (Surface)	21.7

Source: Third National Health Service Survey, 2003

Exposure-Response Functions

Multivariate modeling

A binomial regression model was used to estimate the risk of diarrhea in households with no access to piped water versus those with access after controlling for confounders.

The final model included piped water as the exposure variable (comparing those with piped water to those without) and included the following covariates (confounders):

- *Sanitation.* A binary variable comparing risk of diarrhea in households with a sanitation facility relative to those with none.
- *Income.* A continuous variable showing percent of diarrhea risk change for every unit increase in income.
- *Maternal education.* An ordinal categorical variable comparing risk of diarrhea for the different educational levels relative to the baseline group of no education.

Table 3.8 shows the results of the binomial regression quoting risk ratios (and 95 percent confidence intervals) and their respective P-values for diarrhea.

As the table indicates, the regression analysis found that the risk for diarrhea in households with piped water (safe water proxy) is 0.66 times less than households with no access (i.e., risk is 1.52 if comparing no access to piped water versus access), which is significant at the 5 percent level

Estimating excess annual number of diarrheal episodes in rural China

Given that:

– The crude two-week prevalence for household diarrhea is 2.2 percent;
– The risk for diarrhea in households with no piped water is 1.52 more than households with piped water supply; and
– The proportion of households with no piped water is 0.664.

TABLE 3.7 Socioeconomics and Demographics

Listed Poor Households (n=7,042)	Rural Class (n=7,042)(%)		Income (n=7,042) (%)		Expenditure on Health (n=7,042)(%)	
2.9%	High Economic	18.7	<1000 RMB	26.4	<100 RMB	5.2
	Medium-High	28.1	1000–1699 RMB	34.2	100–299 RMB	27.4
	Medium-Low	35.4	1700–2999 RMB	23.1	300–499 RMB	18.1
	Low Economic	17.8	≥3000 RMB	16.4	≥500 RMB	49.3
Maternal	None	12.4	***Ethnicity***	Han	78.1	
Education	Primary	32.8	*(n=7,028)*	Other	21.9	
(n=6,188)	Secondary	47.7	***Sanitation***	Safe	81.5	
	Higher	7.1	*(n=7,038)*	Unsafe	18.5	

Source: Third National Health Service Survey, 2003.

And if:

$$AF_p = [Pe(RR-1)]/[1+Pe(RR-1)]$$

Where:

AF_p = population attributable fraction
Pe = Prevalence of risk exposure in the population
RR = Relative risk of the outcome for those exposed

Then the population attributable fraction for diarrhea associated with no access to piped water (safe water proxy) can be calculated as follows.

$$AF_p = [0.664(1.52-1)] \\ /1+0.664(1.52-1) \\ = 0.255 \quad (3.1)$$

Under the assumption that the calculated two-week diarrhea prevalence is constant throughout the year, then the annual incidence of household diarrhea in rural China can be estimated as follows:

$$\textit{Household Diarrhea Two-week prevalence} \\ * \textit{No. of fortnights in a year} \quad (3.2)$$

This denotes that, on average, each rural household experiences around 0.6 diarrhea episodes per year (annual incidence) and therefore the total number of diarrhea episodes in rural China can be calculated as follows:

$$\textit{No. of households in rural China with children} \\ \textit{under } 5^{13} \times \textit{Household Diarrhea Incidence} \quad (3.3)$$

Finally, the product of AF_p and the total number of annual household diarrhea episodes in rural China yields the morbidity attributable to lack of piped drinking water.

Estimating Excess Diarrhea Mortality in Children Under 5

The study made certain assumptions to estimate the diarrhea mortality burden in children under five in rural China. It used two different approaches to conservatively calculate the excess total number of diarrhea deaths as a result of lack of access to piped water, which as previously noted has been used as a proxy indicator for poor quality water. The first approach made use of the WHO *Global Burden of Disease* (2002), which presents figures for China, whereas the second approach relied on figures estimated by Jacoby and Wang (2004), who assessed environmental determinants of child mortality in rural China.

TABLE 3.8 Binomial Regression for Risk Ratio for Diarrhea

		Risk Ratio	95% CI	P-value
	Piped Water	0.66	0.44–0.99	0.046
	Sanitation	0.64	0.38–1.06	0.028
	Income	1.05	0.89–1.23	0.150
Maternal Education	Primary	0.84	0.50–1.41	0.513
(baseline group:	Secondary	0.82	0.50–1.35	0.436
No Education)	Higher Education	1.41	0.69–2.89	0.343

Source: Authors' calculations based on Third National Health Service Survey

Estimates based on *Global Burden of Disease* (2002)

Assuming that the AF_p (population attributable fraction) for household diarrhea mortality in children under five equates to that of the diarrhea morbidity as shown in (**1**), then it is possible to estimate the excess number of diarrhea deaths attributable to lack of piped water.

The Global Burden of Disease study, 2002, estimated that nearly 94 percent of diarrhea mortality in the East Asia region lies in the age-group 0–5 years. Therefore, the current study assumed that the same rate applies to China. Using the diarrhea mortality estimates from the Global Burden of Disease, 2002 for China, the total under-5 diarrhea mortality can be calculated as shown below:

$$\text{U5 Diarrhea Mortality} \times \text{Total Diarrhea Deaths} \\ \times \text{Proportion of U5 Dying from Diarrhea}$$

Using the Disease Surveillance Point (DSP) system, Yang et al. (2005) estimated that death rates in rural China were nearly double and triple that in urban China in the years 2000 and 1991 respectively. Taking the conservative assumption that death rates are nearly twice as high in rural than in urban China, we assumed that at least 67 percent of the approximately 102,000 under-5 diarrhea deaths occur in rural China alone.

$$= \textit{Total U5 Diarrhea Deaths} \times \textit{Proportion of Diarrhea Deaths in Rurak China}$$

Finally, the product of AF_p and the total number of diarrhea deaths in rural China yields the attributable mortality count.

Estimates based on Jacoby and Wang (2004)

Jacoby and Wang estimated the impact of access to safe water on under-5 diarrhea mortality probability in rural China for the year 1992. In their study, the definition of safe water incorporated both piped water and deep wells. They estimate that around 0.96 under-five deaths can be averted for every 1,000 live births in the presence of a safe water source.

Since their study reflects 1992 data, adjustments to the averted death rate were conducted to take into account the 2003 U5 mortality rate in China.

$$\textit{Adjusted Death Aversion Rate}$$
$$= \frac{\textit{Aversion Rate Per } 1000 \textit{ Live Births}}{\textit{U5 Mortality Rate in } 1992} \times \textit{U5 Mortality Rate in } 2003$$
$$= \frac{0.96}{33.2} \times 31 = 0.896$$

Using the adjusted death aversion rate and the proposed definition of safe water applied in this current study (piped water only), a more conservative estimate for the number of excess under-five diarrhea deaths in rural China can be calculated as shown below:

$$= \textit{Adjusted Death Aversion Rate} \times \textit{Total Live Births in Thousands Proportion of Diarrhea Deaths in Rural China}$$
$$= \textit{Excess\_deaths}$$

The current study takes the mean of the excess under-5 diarrhea mortality calculated using the above two approaches as a conservative estimate of the excess burden from diarrhea mortality in the U5 population.

Estimating Excess Cancer Mortality

To estimate the excess cancer mortality attributable to water pollution in rural China, we made the following assumptions:

- Rural population size is 782 million (China Statistical Yearbook 2004).
- Approximately 484,840 deaths in rural China in 2003 due to esophageal, stomach, liver, and bladder cancers (see figure 3.7) based on a total of 62 deaths/100,000 population.
- Relative Risk of exposure is 1.56 for those using surface water as a drinking water source versus those relying on well water. This assumption is based on the mean of three measures of effect quoted in a study carried out in Nansi Lake assessing the mortality of stomach, esophagus, and liver cancers comparing populations using lake water against those relying on well water as their drinking water source. This is a very conservative estimate of the RR since two other Chinese studies have quoted ratios as high as 2.44 and 4.52 for overall cancer mortality rates.
- Prevalence of exposure is 21.7 percent (population relying on surface water as their drinking water source, see Table 3.6).

First, we calculate the AFp for cancer mortality in rural China as follows:

$$AF_p = [\mathrm{Pe(OR} - 1)]/[1 + \mathrm{Pe(OR} - 1)]$$

Therefore, the cancer attributable mortality count can be shown as follows:

$AFp \times$ Total Cancer Deaths in Rural China

UNCERTAINTIES AND CAVEATS

The uncertainties in the estimated number of cases of diarrheal morbidity and mortality in rural China due to the absence of piped water supply are mainly related to the following aspects:

- *Limitations with Water Quality Data.* Ideally, household water quality indicators should have been used to reliably assess the true impact of water pollution on diarrhea. However, due to the lack of data, the analysis fell short of doing so and, instead, had to rely on a proxy measure, which assumes that piped water is of high quality. This approach may have introduced some bias (nondifferential misclassification) in exposure ascertainment.
- *Exposure Definition.* Although the analysis takes a conservative approach in defining exposure to safe/unsafe water supply (by considering piped water as the only safe source), it should be emphasized that in rural areas piped water may also fail to meet drinking water standards regularly, hence misclassification of exposure to polluted drinking water may still occur with the chosen exposure metric. This would inevitably affect the estimated exposure-response function and consequently the attributable diarrheal morbidity and mortality counts.
- *Limitations on Inclusion of Confounders for Diarrhea Morbidity.* The present analysis, as mentioned earlier, has been carried out at the household level, with the prevalence calculated based on the presence of one or more diarrhea cases. This approach, although it overcame the problem of clustering, did not allow for the consideration of some important confounders that may affect the estimated exposure-response at the multivariate stage. Age is one such confounder, since those less than 1 year of age probably have a higher risk of disease than those in the other age groups. Another classical confounder that has not been considered for the same reason in the analysis is the effect of sex.
- *Limitations on Cancer Mortality Data.* The assumptions used in the calculation of the attributable cancer mortality count assume that all surface water is polluted, which is not true. The approach used was largely dependent on the results of the Nansi Lake study, which did not deal directly with water quality/pollution, especially for carcinogens but compared lake water consumption to well water, using the former as a proxy indicator for poor quality drinking water. A more rigorous approach would need to consider water quality indicators and assess the proportion of the population exposed not just to a general indicator but more specifically to known carcinogens such as ammonia nitrogen.

Endnotes

1. In China's Surface Water Quality Criteria (Reference Code: GB3838-88), ambient water quality is divided into five categories based on an acidity level (pH) and maximum concentrations for 28 major pollutants. Grades I, II, and III permit direct human contact and use as raw water for potable water systems. Grade IV is restricted to industrial use and recreational uses other than swimming. Grade V is restricted to irrigation. Exceeding the pH or any of the concentration standards for a given grade disqualifies the measured water body from being designated as that grade.
2. The 3rd National Health Service Survey is a household level survey covering about 195,000 households in 95 counties across China prepared every five years by the Health Statistics and Information Center in the Ministry of Health (MOH).
3. The annual report form the Ministry of Water Resources 2004–2005 states: "According to the primary investigation, more than 300 million people in rural areas cannot get safe drinking water."
4. Contamination with organisms causing parasitic diseases and toxins produced by microorganisms (e.g., blue-green algae) are not discussed in this chapter.
5. According to WHO/UNDP (2001), 15 million people in China use drinking water from groundwater wells with arsenic concentration between 0.03–0.65 mg/L (WHO guideline is 0.01 mg/L).
6. In cities, lung cancer is the leading cause.
7. In China, nearly 30 percent of the population lack basic sanitary facilities (China Census Data, 2000, provided by All China Marketing Research Co.Ltd., 2004).
8. http://www.dcp2.org/pubs/DCP/41/
9. Includes both viral and amebic dysentery, of which the first constitutes 93 percent of the cases in Chongqing.

10. Bacteria levels in the different types of water reaching end users (both piped and nonpiped), together with data for the population depending on the different types, were used to estimate the average population-weighted exposure level.
11. WHO, 2006. The annual incidence of typhoid is estimated to be about 17 million cases worldwide See http://www.who.int/water_sanitation_health/diseases/typhoid/en/
12. Report for the national medical and sanitary survey in 19 provinces. http://www.moh. gov.cn/jbkz/index.htm. 2002-10-11.
13. Number of households with children under five obtained from the *China 2000 National Census.*

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Appendix 1 Pollutant-Disease Literature

Reference	Location	Design	Population	Exposure	Disease/Outcome	Measure of Effect (+/– 95% CI)
1. Weyer et al., 2001	Iowa, USA	Retrospective Cohort	• 21,977 women • 55–69 years old	Nitrate in drinking water	Morbidity incidence of: a) Bladder cancer b) Ovarian cancer c) Uterine cancer d) Rectal cancer	• Positive associations for: a) Bladder cancer (across nitrate quartiles, RR=1, 1.69, 1.10, 2.83) b) Ovarian cancer (across nitrate quartiles, RR=1, 1.52, 1.81, 1.84); • Inverse associations for c) uterine cancer d) rectal cancer
2. De Roos et al., 2003	Iowa, USA	Case-control	• 376 adult patients with colon cancer and another 338 with rectum cancer. • 1,244 controls	Nitrate in drinking water	a) Colon cancer b) Rectum Cancer	Weak overall associations of colon or rectum cancers with measures of nitrate in water supply. • Colon cancer OR=1.2 (0.9–1.6) and rectum cancer OR=1.1 (0.7–1.5)
3. Ward et al., 1996	Nebraska, USA	Case-control	• Adult population. • 156 cases and 527 controls.	Nitrate in drinking water	Non-Hodgkin's l ymphoma morbidity	Long-term exposure to water with average nitrate levels of ≥4mg/l was positively associated with risk of NHL, OR=2 (1.1–3.6).
4. Gulis et al., 2002	Trnava district, Slovak Republic	Ecological	Adult population of 237,000 people.	Nitrate in drinking water	• Cancer of: a) Stomach b) Colorectal c) Bladder d) Kidney • Non-Hodgkin lymphoma.	• For all cancer in both males and females, Standardized Incidence Ratios increased from villages with low to medium to high levels of nitrate (P for trend <0.001)

(*continued*)

Appendix 1 (continued) Pollutant-Disease Literature

Reference	Location	Design	Population	Exposure	Disease/Outcome	Measure of Effect (+/– 95% CI)
						• This pattern in the SIRs (from low to high nitrate level) was also seen for stomach cancer in women (0.81, 0.94, 1.24; P for trend= 0.10), colorectal cancer in women (0.64, 1.11, 1.29; P for trend <0.001) and men (0.77, 0.99, 1.07; P for trend= 0.051), and non-Hodgkin lymphoma in women (0.45, 0.90, 1.35; P for trend= 0.13) and men (0.25, 1.66, and 1.09; P for trend=0.017). – There were no associations for kidney or bladder cancer.
5. Chen et al., 1986	Southwestern Taiwan	Matched case-control	204 cancer deceased cases and 368 matched alive community controls	High arsenic levels in drinking water	Bladder, lung and liver cancer	– Positive dose-response relationship observed. – Age-sex adjusted odds ratios were 3.90, 3.39, and 2.67 for bladder, lung and liver cancer respectively.
6. Ahsan et al., 2006	Bangladesh	Cohort	- Adult population - 11,746 subjects	Arsenic in drinking water	Premalignant skin lesions	– Dose-response relationships for increasing arsenic were associated with skin lesions.

						– Adjusted odds ratios of 1.91 (1.26–2.89), 3.03 (2.05–4.5), 3.71 (2.53–5.44), and 5.39 (3.69–7.86) reported with increasing doses.
7. Mazumder et al., 2005	West Bengal, India	Case-control	- 108 cases with arsenic-caused skin lesions and 150 controls	Arsenic in drinking water	Bronchiectasis morbidity in persons with skin lesions from arsenic exposure	– Subjects with arsenic-caused skin lesions had a 10-fold increased prevalence of bronchiectasis compared with those without. – Adjusted OR = 10 (2.7–37)
8. Rahma, et al., 1999	Bangladesh	Analytical Cross-sectional	- Adult population 30 years old or more - 1595 subjects	Arsenic in drinking water	Hypertension	Dose response relationships were established for different levels of exposure to arsenic and adjusted for age, sex and body mass index. The prevalence ratios for exposed/unexposed were 1.2, 2.2, 2.5 for exposure categories <0.5 mg/L, 0.5 to 1.0 mg/L, & >1.0 mg/L, and 0.8, 1.5, 2.2, 3.0 for exposure categories <1.0 mg-y/L, 1.0 to 5.0 mg-y/L, >5.0 but </=10.0 mg-y/L, and >10.0 mg-y/L.
9. Tseng et al., 1996	Taiwan	Analytical Cross-sectional	- 582 Adult - Ages between 40 and 60	Arsenic in drinking water	Peripheral vascular disease	OR=2.77 (0.84–9.14) and 4.28 (1.26–14.54) for arsenic exposure of 0.1–19.90mg/l-years and 20.0 and more mg/l-years respectively compared to those who were not exposed.

(*continued*)

Appendix 1 (continued) Pollutant-Disease Literature

Reference	Location	Design	Population	Exposure	Disease/Outcome	Measure of Effect (+/– 95% CI)
10. Aschengrau et al., 1989	USA, Boston	Case-control	286 women who had spontaneous abortion through week 27 of gestation and 1,391 controls	Arsenic and mercury in drinking water	Spontaneous abortion	– In the adjusted analysis high levels of arsenic and detectable levels of mercury were associated with increases in the risk of spontaneous abortion, OR 1.5 (0.4–4.7) and OR 1.5 (1.0–2.3) respectively.
11. von Ehrenstein et al., 2006	West Bengal, India	Cohort	202 married women	Arsenic in drinking water	Stillbirths and neonatal deaths among other pregnancy-related outcomes	Exposure to arsenic levels of 200 microg/L or higher was associated with: OR=6.07 (1.54–24, p=0.01) for stillbirth; OR=2.81 for neonatal death.
12. Rahman et al., 2006	Bangladesh	Case-control	– Adult population – 163 cases with keratosis (taken as exposed to arsenic) and 854 unexposed controls	Arsenic in drinking water	Diabetes mellitus	Crude prevalence ratio for DM among exposed was 4.4 (2.5–7.7) and increased to 5.2 (2.5–10.5) after adjusting for age, sex, and BMI.
13. Mazumder et al., 2000	West Bengal, India	Case-control	7,683 participants of all ages	Arsenic in drinking water	Respiratory problems	Age-adjusted odds ratio estimates for cough were 7.8 (3.1–19.5) for females and 5.0 (2.6–9.9) for males; for chest sounds OR for females was 9.6 (4.0–22.9) and for males 6.9 (3.1–15.0). OR for shortness of breath in females was 23.2 (5.8–92.8) and in males 3.7 (1.3–10.6).

14. Mazumder 2005	West Bengal, India	Cross-sectional	7,863 people residing in arsenic-affected districts	Arsenic in drinking water	Hepatomegaly	– RR for **hepatomegaly** was 3.41 ($p<0.001$). – The incidence of **hepatomegaly** was found to have a linear relationship proportionate to increasing arsenic exposure in drinking water in both sexes ($p<0.001$).
15. Aschengrau et al., 1993	Massachusetts, USA	Case-control	– 1,039 congenital anomaly cases, 77 stillbirths, 55 neonatal deaths, and 1,177 controls	Lead in drinking water (among other pollutants)	– Stillbirths – Cardiovascular defects – Other late adverse pregnancy outcomes	For women exposed to detectable lead levels: – Frequency of stillbirths increased OR=2.1 (0.6–7.2) – Frequency of cardiovascular defects increased OR=2.2 (0.9–5.7).
16. Chandrashekar et al., 2004	Karnataka, India	Analytical cross-sectional	1,131 children, 12–15 year old.	Fluoride in drinking water	Dental fluorosis	There was a significant positive linear correlation ($r=0.99$) between community fluorosis index (CFI) and water fluoride level. Lowest fluoride levels in drinking water (0.22 ppm) corresponded to 13.2% prevalence in dental fluorosis and highest (3.41 ppm) corresponded to 100% prevalence.
17. Khandare et al., 2005	Bihar state, India	Case-control	– Young children – 240 cases in village with high F (HFV), and 1,443 in control village	Fluoride in drinking water	Severe dental deformities	Dental mottling was observed in 50% and skeletal deformities in 20% of children in HFV vs. 1% and 0% respectively in control villages.

(continued)

Appendix 1 (continued) Pollutant-Disease Literature

Reference	Location	Design	Population	Exposure	Disease/Outcome	Measure of Effect (+/– 95% CI)
18. Moe et al., 1991	Cebu, Philippines	Cohort	690 under-2-year-olds exposed to different levels of bacterially contaminated water	Bacterial indicators in drinking water (fecal coliforms, E.coli, enterococci)	Diarrheal illness in children under 2	Children drinking water with greater than 1,000 E.coli per 100 ml had significantly higher rates of diarrheal disease than those drinking less contaminated water (RR= 1.7, p=.002).
19. Garg et al., 2005	Ontario, Canada	Cohort	1958 adults (includes 675 asymptomatic, 909 moderate symptoms of acute self limited gastroenteritis, and 374 severe symptoms of gastroenteritis) followed up after an outbreak	Bacteria-contaminated drinking water	Risk of hypertension and reduced kidney function after acute gastroenteritis.	After a mean follow-up of 3.7 years, hypertension was diagnosed in 27% of participants w/ no symptoms, 32.2% w/ moderate symptoms and 35.9% in those with severe symptoms (trend p=0.009). RR for hypertension for mod and severe is 1.15 and 1.28. Similar graded association found for reduced kidney function (trend p=0.03).
20. Robins-Browne et al., 2004	Melbourne, Australia	Case-control	696 patients with gastroenteritis and 489 controls	E-coli contaminated drinking water	Gastroenteritis	Atypical enteropathogenic E.coli (EPEC) were significantly higher in patients with gastroenteritis (12.8%) than asymptomatic persons (2.3%). RR= 5.57, p<0.0001.

21. Ashraf et al., 1997	Aligarh District of Uttar Pradesh, India	Cohort	1,270 persons in households with either standpost water supply or piped water supply	Bacteria-contaminated drinking water	Waterborne diseases of bacterial origin: a) Typhoid b) Bacillary dysentery c) Diarrhea	– RR for overall morbidity in standpost vs. piped water =1.7 – RR for typhoid= 1.57 – RR for bacillary dysentery= 1.23 – RR for diarrhea= 1.2
22. Ries et al., 1992	Piura, Peru	Matched case-control	50 cases and 100 matched controls	Poor water quality (insufficiently chlorinated and contaminated with fecal coliform bacteria)	Epidemic cholera	– Cholera was associated with drinking unboiled water OR=3.9 (1.7–8.9).
23. Swerdlow et al., 1992	Trujillo, Peru (second largest city)	Matched case-control	46 cases and 65 symptom free and serologically uninfected controls	Poor water quality (insufficiently chlorinated and contaminated with fecal coliform bacteria)	Epidemic cholera	Cholera was associated with: – Drinking unboiled water OR=3.1 (1.3–7.3) – Drinking from a household water storage container in which hands had been introduced into the water OR=4.2 (1.2–14.9)

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4

Valuation of Environmental Health Risks

This chapter provides an overview of methods used by economists to value morbidity and premature mortality and uses them to quantify the health damages associated with outdoor air pollution and water pollution. An important goal of the CECM/VEHR project was to contribute to the literature on health valuation in China. This chapter summarizes the results of original studies conducted in Shanghai and Chongqing to estimate people's willingness to pay to reduce risk of premature death. The chapter also discusses the adjusted human capital (AHC) approach—the official approach used to value mortality risks in China. The excess deaths associated with PM_{10}, when monetized using the AHC approach, total approximately 0.8 percent of GDP; when valued using the best estimate of the VSL based on studies in Shanghai and Chongqing (1.0 million Yuan), they reach 2.9 percent of GDP. Total health costs associated with air pollution are 1.2 percent (using AHC) and 3.8 percent (using VSL) of GDP. Premature mortality constitutes approximately three-quarters of the total monetized health costs of air pollution, using either approach.

The purpose of this chapter is to attach monetary values to the health effects associated with air and water pollution in Chapters 2 and 3 of this report. Physical measures of health damages are useful indicators of the costs of environmental degradation, and hence of the benefits that would accrue if air pollution were reduced. However, it is often useful to measure environmental damages in monetary terms. This chapter describes the methods used to monetize health effects and ends by presenting the dollar value of the health damages associated with air pollution and water pollution that were quantified in Chapters 2 and 3.

The chapter begins with a brief review of the economic concepts behind valuing human health effects. We then discuss how we derived the unit values applied in this analysis, followed by the results of the analysis.

VALUING MORTALITY RISKS

In benefit-cost analyses of environmental programs conducted in the United States and the European Union, mortality risks are typically valued using the "value of a statistical life" (VSL)—the sum of what people would pay to reduce their risk of dying by small amounts that, together, add up to one statistical life. When estimates of the VSL are unavailable, the human capital approach (foregone earnings) is often used to place a lower bound on the VSL. In valuing the premature deaths associated with environmental degradation in China, we used both approaches: the VSL, estimated from original stated preference studies conducted in Shanghai and Chongqing, and the adjusted human capital (AHC) approach.

The Willingness to Pay (WTP) Approach

The reductions in premature mortality presented in Chapter 2 describe the number of statistical lives lost due to air pollution. In reality, decreases in air pollution reduce the risk of dying over a stated period for each person exposed to air pollution. If the risk of death is reduced by 1 in 10,000 annually for each of 10,000 people exposed to air pollution, then on average one life—termed a statistical life—will be saved.

Because programs to reduce air pollution reduce the risk of dying for each person in the exposed population, it is these risk reductions that are valued. The willingness-to-pay approach to valuing reductions in the risk of death values each risk reduction by what a person would pay to obtain it. For example, a person might be willing to pay 200 Yuan to reduce his/her risk of dying by 1 in 10,000 during the coming year. This is his/her value of the risk reduction. By definition, the value of a statistical life is the sum of individuals' willingness to pay for small risk reductions that together add up to one statistical life. If a reduction in air pollution reduces each person's risk of dying by 1 in 10,000, it will save one statistical life in a population of 10,000. The amount that the 10,000 people together would pay for the risk reduction is known as the value of a statistical life (VSL). If each of 10,000 people were willing to pay 200 Yuan, the VSL = 10,000 × 200 Yuan = 2 million Yuan.

Approaches to Measuring the VSL

In practice, how do we know what people are willing to pay for a 1-in-10,000 risk reduction? Internationally, this is usually estimated from compensating wage differentials in the labor market, or from contingent valuation surveys in which people are asked directly what they would pay for a reduction in their risk of dying. The basic idea behind compensating wage differentials is that jobs can be characterized by various attributes, including risk of accidental death. If workers are well-informed about risks of fatal and non-fatal injuries, and if labor markets are competitive, riskier jobs should pay more, holding worker and other job attributes constant. In order to estimate compensating wage differentials empirically, an equation is estimated to explain variations in the wage received by workers as a function of worker characteristics (age, education, human capital) and job characteristics, including risk of fatal and non-fatal injury (Viscusi 1993). In theory, the impact of a small change in risk of death on the wage should equal the amount a worker would have to be compensated to accept this risk. For small risk changes, this is also what the worker should pay for a risk reduction.

For the compensating wage approach to yield reliable estimates of the VSL, it is necessary that workers be informed about fatal jobs risks and that there be sufficient competition in labor markets for compensating wage differentials to emerge. To measure these differentials empirically requires accurate estimates of the risk of death on the job—ideally, broken down by industry and occupation. The researcher must also be able to include enough other determinants of wages that fatal job risk does not pick up the effects of other worker or job characteristics. For example, since data on risk of injury are usually collected at the industry level, it is important to control adequately for other sources of inter-industry wage differentials.[1]

Empirical estimates of the value of a statistical life based on compensating wage studies conducted in the U.S. lie in the range of \$0.6 million to \$13.5 million (1990 USD) (Viscusi 1993; USEPA 1997). For Taiwan, Liu et al. (1997) report a VSL of \$413,000 (1990 USD); Liu and Hammitt (1999) report a VSL of \$650,000. However, similar studies have not been conducted in mainland China. It should be emphasized that the average age of workers in compensating wage studies is usually around 40 years of age and that the risks assumed in the labor market are, to some degree, voluntarily borne. Both of these points pose difficulties in using compensating wage differentials to value changes in environmental risks. If risk of death due to air pollution is proportional to baseline risk of death, as is assumed in Pope et al. (1995, 2002), 59 percent of the statistical lives saved by reductions in particulate matter in China are estimated to accrue to persons over the age of 65 (Appendix Table A.1). To the extent that older people with fewer years of life remaining would pay less to reduce mortality risks, compensating wage differentials may overstate the value of their statistical lives. The fact that environmental risks are involuntarily borne, however, argues that compensating wage differentials, with all other things remaining equal, may understate the value of environmental risk reductions.

Problems with compensating wage differentials suggest that it may be worthwhile to use direct questioning approaches when valuing changes in life expectancy, since they can be tailored to the age at which risk reductions occur and to the nature of the risks valued. Contingent valuation (CV) studies have both advantages and disadvantages. One advantage of a contingent valuation study is that is easier to see how WTP for a risk reduction varies with age and income. A disadvantage of CV studies is that they often make apparent respondents' difficulties in consistently valuing small probabilities.[2]

Because contingent valuation studies ask hypothetical questions, it is standard practice for these studies to include tests of internal and external validity of responses. External scope tests vary the size of the risk reduction valued across respondents to see whether WTP increases with the size of the risk reduction. Failure of WTP to increase with the size of the risk reduction suggests that respondents do not perceive risk changes correctly, or are valuing a generalized commodity ("good health") rather than a quantitative risk reduction. Internal scope tests check to see whether WTP increases with the size of the risk reduction for a given respondent. Tests of external validity also include checking whether responses vary, as expected, with income.

To our knowledge, three contingent valuation studies have been conducted in China to value quantitative reductions in risk of death: Hammitt and Zhou (2005); Wang and Mullahy (2006); and Zhang (2002). The VSLs obtained in these studies, based on mean WTP, are listed in Table 4.1. VSLs range from 250,000 to 1.7 million Yuan, depending on the study and model used to fit the data. Only one of these studies performed an external scope test (Hammitt and Zhou 2005). Unfortunately, respondents' WTP for reductions in risk of death failed to respond to the size of the risk change. For this reason, and in order to estimate the WTP of older persons, original studies were conducted in Shanghai and Chongqing to complement the environmental cost modeling. The details of these studies, conducted by Krupnick et al. (2006), are reported in an Annex and in Box 1.

TABLE 4.1 Estimates of the Value of a Statistical Life in Chinese Studies

Study	Million Yuan
Wang and Mullahy (2006)	0.3–1.25
Zhang Xiao (2002)	0.24–1.7
Hammitt and Zhou (2005)*	0.26–0.51
Krupnick et al. (2006)	1.4

Source: Authors calculation.

Choice of WTP Values for the ECM

In valuing premature mortality due to air pollution, we use the preferred VSL reported by Krupnick et al. (2006), 1.4 million Yuan, based on pooled data from Shanghai and Chongqing, but adjusted to reflect differences in income between Shanghai, Chongqing, and the rest of China. Once the income adjustment is made, the Krupnick et al. (2006) figure is approximately 1 million Yuan.[3] We note that this falls within the range of values reported in the other studies listed in Table 4.1. Following the practice used in the U.S. and Europe, we apply the same value to all lives lost due to air pollution, regardless of location (i.e., of per capita GDP). This practice is followed in the United States for political, rather than economic, reasons.

As noted above, a key reason for conducting an original valuation study was to examine how WTP

BOX 4.1 The Willingness to Pay for Mortality Risk Reductions in Shanghai and Chongqing

One goal of the valuation of the environmental health risk (VEHR) component of the project was to estimate the value of reducing risks of death by conducting contingent valuation surveys in Shanghai and Chongqing. The surveys were conducted in the winter and summer of 2005, respectively, with a second survey in Shanghai in the spring of 2006. The survey questionnaire, with minor changes, was identical to those administered in the U.S., Canada, U.K., France, Italy, and Japan by Alberini et al. (2004). The target population comprised persons 40 to 80 years old. Respondents were asked how much they would pay over the next 10 years for a product that would reduce their risk of dying, over the 10-year period, by 10 in 1,000 and by 5 in 1,000 (i.e., by 1 in 10,000 and 5 in 10,000 per year). Bids were elicited by either a double-bounded dichotomous choice (DC) method or a payment card (PC). The questionnaire was self-administered on a computer with voiceovers.

Samples, stratified by community and neighborhood, were drawn at random in each city. In Shanghai, 1,920 persons were initially contacted and invited to take the DC survey, and 1,224 participated, an acceptance rate of 64 percent. Another 600 accepted the payment card version of the survey. In Chongqing, 1,250 persons were contacted and invited to take the survey; 1,067 enrolled, a response rate of 85.4 percent.

The results show that respondents care very much about reducing their mortality risks, and are willing to pay for this. Indeed, the mean VSLs—using the same estimation approach as was used for the countries listed above—are in the same range as the other countries, in PPP terms. Using a conservative estimation approach gives a mean VSL of 1.4 million yuan when data are pooled for the 5 in 10,000 annual risk reduction from the DC versions of the survey. When data from the two cities are analyzed separately, the Chongqing VSL is slightly lower than that for Shanghai, but by much less than would be suggested by income differences. The VSL estimates for the DC and PC methods in Shanghai are not statistically different from one another.

The results pass some validity tests and not others. The external scope test (in which the WTP for a 5-in-10,000 risk reduction by one group is compared to that of a 10-in-10,000 risk reduction by another group) was passed by the general population using the PC method, but only by highly educated people in Chongqing using the DC method. The regression results are reasonably intuitive and conform to expectations. For instance, those persons with more income, more education, and who are in poorer health are willing to pay more for the risk reduction.

One concern is that a large fraction of respondents had to be eliminated from each of the analyses because of various problems with their WTP answers, such as illogical responses.

for mortality risk reductions varied with the age of the respondent, and to examine the WTP of older persons.[4] In the international literature, there is some weak evidence that WTP for mortality risk reductions falls later in life. Alberini et al. (2004), based on surveys similar to Krupnick et al. (2006) conducted in the U.S. and Canada, find that the WTP of persons over 70 is approximately 25 percent lower than the WTP of persons 40 to 69. The results of the survey work in China on this point are mixed. When WTP from two samples of respondents in Shanghai and Chongqing is analyzed as a function of covariates (Krupnick et al. 2006, Table 18), WTP is approximately 28 percent lower for persons over 65 than for persons below that age, other things being equal. This result, however, is not supported by analysis of a third sample of respondents in Shanghai. In light of this evidence, we do not allow the VSL to vary with age in the tables below.

A related problem occurs in valuing the lives of children. Chapter 3 estimates that significant numbers of deaths among children under the age of 5 would be avoided if rural households had better access to water and sanitation. Valuing children's deaths is problematic using the willingness to pay approach. Children are not thought to have well-defined WTPs, so it is parents' WTP for reduced risks to their children that is usually measured.[5] The USEPA does not believe that there are enough such studies to use a separate estimate of the VSL for children. We follow this approach and apply the VSL estimated by Krupnick et al. (2006) to value premature mortality in children.

The Adjusted Human Capital Approach

An alternate approach to willingness to pay is to use the productivity loss associated with premature mortality (i.e., forgone earnings) to value loss of life. This values an individual by what he produces and assumes that this value is accurately measured by his earnings. The adjusted human capital (AHC) approach, which is widely used in China, represents an important departure from the traditional human capital approach. Because the use of foregone earnings would assign a value of zero to the lives of the retired and the disabled, the AHC approach avoids this problem by assigning the same value—per capita GDP—to a year of life lost by all persons, regardless of age. For this reason, the adjusted human capital approach can be viewed as a social statement of the value of avoiding premature mortality.

In practice, the AHC values a life lost at any age by the present discounted value of per capita GDP over the remainder of the individual's expected life. In computing the AHC measure, real per capita GDP is assumed to grow at rate α annually and is discounted to the present at the rate r. Adjusted human capital, HC_m, is thus given by (4.1)

$$HC_m = GDP_{pc0} \sum_{i=1}^{t} \frac{(1+\alpha)^i}{(1+r)^i} \qquad (4.1)$$

where GDP_{pc0} is per capita GDP in the base year and t is remaining life expectancy. In the base case calculations $\alpha = 7\%$ and $r = 8\%$.

Equation (4.1) implies that HC_m will vary with the age of the person who dies and will vary by city or province, assuming that per capita GDP varies by city or province. Remaining life expectancy, which does not vary by province in the published data, is calculated using Chinese life tables assuming that the age distribution of deaths due to air pollution is identical to the age distribution of deaths due to respiratory and cardiovascular diseases. As shown in the appendix, the average number of life-years lost due to air pollution is approximately 18. Per capita GDP in the base year (2003) differs by city. Table 4.2 shows the adjusted human capital measure computed for different cities, assuming r = 8% and allowing α to equal 6%, 7%, and 8%. The central case estimates below correspond to HC_m in the second column from the right.[6]

VALUING MORBIDITY

In principle, economists value avoided morbidity by the amount a person will pay to avoid (the risk of) an illness, just as risk of death is valued by what people will pay to reduce it. In the case of morbidity, WTP should capture the value of the pain and suffering avoided, as well as the value of time lost due to illness (both leisure and work time) and the costs of medical treatment. If some of these costs are not borne by the individual, and are therefore not reflected in his willingness to pay, the value of the avoided costs must be added to WTP to measure the social benefits of reduced morbidity.

In cases where WTP estimates are not available, analysts often rely on cost-of-illness (COI) estimates as a lower bound to the theoretically correct value of avoiding illness. Cost-of-illness studies estimate the lost earnings associated with chronic illness that result both from reduced labor force participation and lower earnings conditional on participation (Bartel and Taubman 1979; Krupnick and Cropper 2000), and add to these medical costs associated with the disease. The COI is a lower bound to WTP because it ignores the value of pain and suffering associated with illness and the value of lost leisure time. In regulatory impact analyses of air pollution regulations published by the U.S. Environmental Protection Agency (USEPA 1997), it is often the case that coronary heart disease and stroke are valued using cost-of-illness estimates, as WTP estimates are unavailable.

In the ECM, we approximate WTP for chronic bronchitis using benefits-transfer methods.

TABLE 4.2 Adjusted Human Capital (HC_m) of Different Cities with Different Growth Rates of Per Capita GDP (Base year: 2003)

Growth Rate of GDP/Capita (α, %)		6	7	8
Discount Rate (r, %)		8	8	8
	Value	15.14	16.50	18
$\sum_{i=1}^{t}[(1+\alpha)/(1+r)]^i$	Hypothesis of α ∇ 7% Being 1	0.92	1	1.09
Cities	Per Capita GDP/Yuan	HC_m (10,000 Yuan)		
Beijing	32,061	48.55	52.89	57.71
Tianjin	26,532	40.18	43.77	47.76
Shijiazhuang	15,188	23.00	25.06	27.34
Taiyuan	15,210	23.03	25.09	27.38
Huhehaote	18,791	28.45	31.00	33.82
Shenyang	23,271	35.24	38.39	41.89
Dalian	29,206	44.22	48.18	52.57
Changchun	18,705	28.32	30.86	33.67
Haerbin	14,872	22.52	24.53	26.77
Shanghai	46,718	70.74	77.07	84.09
Nanjing	27,307	41.35	45.05	49.15
Hangzhou	32,819	49.70	54.14	59.07
Ningbo	32,639	49.42	53.84	58.75
Hefei	10,720	16.23	17.68	19.30
Fuzhou	20,520	31.07	33.85	36.94
Xiamen	35,009	53.01	57.75	63.02
Nanchang	14,382	21.78	23.73	25.89
Jinan	23,590	35.72	38.92	42.46
Qingdao	23,398	35.43	38.60	42.12
Zhengzhou	17,063	25.84	28.15	30.71
Wuhan	21,457	32.49	35.40	38.62
Changsha	14,810	22.43	24.43	26.66
Guangzhou	48,372	73.25	79.80	87.07
Shenzhen	54,545	82.59	89.98	98.18
Nanning	7,874	11.92	12.99	14.17
Haikou	16,730	25.33	27.60	30.11
Chongqing	8,077	12.23	13.32	14.54
Chengdu	18,051	27.33	29.78	32.49
Guiyang	10,962	16.60	18.08	19.73
Kunming	16,312	24.70	26.91	29.36
Xian	12,233	18.52	20.18	22.02
Lanzhou	14,540	22.02	23.99	26.17
Xining	7,110	10.77	11.73	12.80
Yinchuan	11,788	17.85	19.45	21.22
Wulumuqi	19,900	30.13	32.83	35.82

Source: Authors calculations.

For hospital admissions, we rely on cost-of-illness estimates.

Estimating WTP to Avoid Chronic Bronchitis

In the case of common illnesses, such as diarrheal disease, economists usually try to value reductions in days of illness, treated as certain. For illnesses that are rarer, such as chronic bronchitis, it is appropriate to view exposure to pollutants as increasing the risk of serious illnesses and to value reductions in risk of illness.

To value reductions in the risk of chronic bronchitis, one could ask individuals directly what they would pay to lower their risk of experiencing these conditions. An alternate approach that has proved successful (Viscusi, Magat, and Huber 1991) is to ask individuals to make trade-offs between the risk of contracting a serious illness and the risk of death (e.g., dying in an auto accident). These risk-risk trade-offs establish an equivalence between the utility of good health and the utility of the disease. For example, in a U.S. study involving trade-offs between risk of contracting chronic bronchitis and risk of dying in an auto accident, people's choices implied that the utility of living with chronic bronchitis was about 0.68 of the utility of living in good health (Viscusi, Magat, and Huber 1991). If good health is scaled to equal 1 and death scaled to equal 0, then this is equivalent to saying that living a year with chronic bronchitis is equal to losing 0.32 of a year of life. This number can be converted to the value of a statistical case of chronic bronchitis by multiplying the value of a statistical life by 0.32.

The risk-risk tradeoff approach is closely related to methods used in the public health literature to establish QALY weights for chronic disease—the ratio of the utility of living with the disease to the utility of living in good health (Miller, Robinson, and Lawrence 2006).[7] It is therefore possible to draw on the QALY literature to establish the fraction of a year lost if one has chronic bronchitis. Clearly this equivalence will depend on the severity of the case of chronic bronchitis. It is, therefore, not surprising that the QALY weights reported in the literature for chronic bronchitis vary widely.

Although one attempt has been made to estimate a QALY weight for chronic bronchitis in China, we choose a value from the international literature. In survey work in China, Hammitt and Zhou (2005) use both risk-risk tradeoffs and standard gambles to determine the utility lost due to chronic bronchitis. However, the case of chronic bronchitis they describe is a very mild one. We therefore appeal to the international literature on QALY weights for chronic bronchitis, and select a value in the middle of the range of weights reported by Miller, Robinson, and Lawrence (2006, Appendix A). Specifically, we assume that living a year with chronic bronchitis is equivalent to losing 0.4 years of life.

When excess deaths are valued using the VSL from Krupnick et al. (2006), the value of a statistical case of chronic bronchitis is computed as 0.4 $*$ VSL. When the AHC approach is used to value excess deaths, we compute HC_m using the expected number of years a person will live with chronic bronchitis in place of t in equation (1) and multiply the result by 0.4.

Valuing Hospital Admissions

For most acute illness episodes (restricted activity days, asthma attacks), contingent valuation is the method most often used to value avoided morbidity (Loehman and De 1982; Freeman 1993). In China, few contingent valuation studies have been conducted to value acute illness. Notable exceptions are Hammitt and Zhou (2005), who estimate WTP to avoid a cold in Anqing and Beijing, and studies conducted in Taiwan to estimate WTP to avoid a recurrence of acute respiratory illness (Alberini et al. 1997). Unfortunately, we know of no studies that estimate WTP to avoid a respiratory or cardiovascular hospital admission. We therefore use the cost-of-illness approach to value hospital admissions.

TABLE 4.3 Illness Costs for Hospital Admissions in China in 2003 (Yuan/episode)

	Direct Plus Indirect Costs			
Cause of Admission	Large-Scale City	Middle-Scale City	Small-Scale City	Indirect Cost
Respiratory	8,474	5,071	2,593	514
Cardiovascular	12,326	8,506	6,028	514

Source: Authors calculations based on the China National Health Survey 2003.

National surveys on health services were carried out in China in 1998 and 2003 in which medical costs were reported. The 1998 survey provided disease-specific medical cost information, whereas the 2003 survey only provided all-disease average costs. However, the 2003 report calculated the increase in average medical cost from 1998 to 2003. Assuming that each disease-specific cost increased by the same proportion, we estimate the disease-specific costs in 2003, as shown in Table 4.3. The direct costs of illness include all the costs in hospital, including expenditures for medical examinations, drugs, and therapy, as well as the cost of the hospital stay. Indirect costs include the patient's time lost from work, as well as the work-days lost by patients' families. In China, it is common that the family, colleagues, or friends of the patients leave their work to visit the patients in hospital. The economic loss from this kind of work absence has been valued as well. Illness costs are broken down by city size, as well as type of hospital admission.

MONETARY HEALTH COSTS OF AMBIENT AIR POLLUTION

Tables 4.4 and 4.5 summarize the monetary costs of ambient air pollution. Table 4.4 summarizes the costs of ambient air pollution using the AHC approach to value both premature mortality and chronic bronchitis. Table 4.5 repeats the calcu-

TABLE 4.4 Health Costs Associated with Outdoor Air Pollution in China, 2003 Adjusted Human Capital Approach (Bil. Yuan)

		Morbidity			
Estimate	Excess Deaths	Chronic Bronchitis	Direct Hospital Costs	Indirect Hospital Costs	Total Costs
95th %ile	178.7	47.7	4.82	0.670	231.8
Mean	110.9	42.5	3.41	0.470	157.3
5th %ile	35.8	36.9	1.88	0.264	74.9

Source: Authors calculations based on the China National Health Survey 2003 and other sources.

lations using the VSL to monetize premature mortality and chronic bronchitis. The mean estimates and 5th and 95th percentiles refer to the uncertainty bounds for the number of cases of mortality and morbidity.

Several points are worth noting. The mean total health cost associated with ambient air pollution in urban areas of China in 2003 is 157 billion Yuan if the adjusted human capital approach to valuation is used, and 520 billion if WTP estimates from the VEHR study are used. Use of WTP increases total costs by a factor of 3.3, bringing health costs to 3.8 percent of 2003 GDP. Using the AHC approach, health costs are still 1.2 percent of GDP. As in many studies, the damages associated with premature mortality dominate the total: they are 71 percent of health costs using the AHC approach and 76 percent using the WTP approach. However, in both cases chronic bronchitis costs are significant—over 20 percent of total costs.

TABLE 4.5 Health Costs Associated with Outdoor Air Pollution in China, 2003 Willingness to Pay Approach (Bil. Yuan)

		Morbidity			
Estimate	Excess Deaths	Chronic Bronchitis	Direct Hospital Costs	Indirect Hospital Costs	Total Costs
95th %ile	641.1	136.7	4.82	0.670	783.3
Mean	394.0	122.1	3.41	0.470	519.9
5th %ile	135.6	106.2	1.88	0.263	243.9

Source: Authors calculations based on the China National Health Survey 2003 and other sources.

MONETARY HEALTH COSTS OF WATER POLLUTION

Chapter 3 quantifies three health endpoints associated with water pollution: excess cases of morbidity associated with diarrheal disease in children under 5 years, premature mortality associated with diarrheal disease in this age group, and premature mortality due to cancers of the digestive system. Here we monetize the premature mortality associated with water pollution and the morbidity associated with diarrheal disease. Cancer morbidity is not monetized due to the difficulty in calculating the cost of treating an episode and the percent of episodes treated. For this reason, the estimates below must be regarded as lower bounds to the total health costs associated with water pollution in China.

To monetize premature mortality using the AHC approach requires an estimate of rural per capita GDP. There are no official data on urban and rural GDP in China. However, based on our calculations GDP is approximately 5,384Yuan.[10] We assume, following the Green National Accounting Report, that premature mortality due to cancers of the digestive system results in a loss of 21 years of life. This implies, for the central case of $\alpha = .07$ and $r = .08$, that $HC_m = 102{,}242$ Yuan for a statistical cancer death. Table A.1 implies a loss of approximately 78 years of life for a child who dies of diarrheal disease before age 5. Using the same per capita rural GDP figure implies that $HC_m = 297{,}251$ yuan for a statistical death due to diarrheal disease. Using the VSL approach, both deaths are valued at 1.0 million Yuan. These assumptions lead to the results reported in Table 4.6.

TABLE 4.6 Health Costs Associated with Water Pollution in China, 2003 (Bil. Yuan)

Disease	Morbidity Cost	AHC Mortality Cost	VSL Mortality Cost
Diarrhea	0.22	4.16	14.0
Cancer	N/A	5.31	52.0
Total		9.47	66.0

Source: Authors Calculations

Morbidity due to diarrheal disease in children under the age of 5 is valued at a cost of two days of caregivers' time. This was calculated as 29.5 Yuan per case, the pro-rated value of per capita rural GDP.[11]

Although an underestimate of the total health costs of water pollution, the costs in Table 4.6 are about an order of magnitude smaller than the health costs associated with outdoor air pollution. This is true even when outcomes are valued using the same VSL for persons in rural and urban areas. Compared to the health cost of air pollution, the health cost of water pollution are relatively low. This does not, of course, mean that individual projects to improve rural drinking water quality will necessarily yield smaller net benefits than specific projects to improve urban air quality. It should also be noted that improvements in surface water quality, which are one way of reducing the costs of drinking water treatment, will yield non-health as well as health benefits.

Endnotes

1. Estimates of compensating wage differentials are often quite sensitive to the exact specification of the wage equation. Black et al. (2003), in a reanalysis of data from U.S. compensating wage studies requested by the USEPA, conclude that the results are too unstable to be used for policy.
2. For example, WTP for a reduction in risk of death seldom increases in proportion to the size of the risk change, which suggests that respondents do not perceive risk changes as economists expect them to.
3. This adjustment is made using the ratio of average disposable income in China to average disposable income in Shanghai and Chongqing. The income elasticity of 0.48 from Krupnick et al. (2006) is used to make the adjustment.
4. For this reason, over one-third of sample respondents were chosen to be 60 years of age and older.
5. For a summary of this literature see the USEPA's *Handbook on Valuing Children's Health* at http://yosemite.epa.gov/ee/epa/eed.nsf/webpages/HandbookChildrensHealthValuation.html
6. It should be noted that the adjusted human capital values in Table 4.2 pertain to cities, whereas the results below are reported for provinces and municipalities.
7. One such approach is the standard gamble approach, used by Hammitt and Zhou (2005). This approach asks a person, were he to contract chronic bronchitis, what risk of death ρ he would accept to undergo an operation that would cure the disease with probability $1-\rho$.
8. Total Health Costs = Cases of Premature Mortality * Cost per Case + Cases of Chronic Bronchitis * Cost per Case + Direct Cost of Hospital Admissions + Indirect Cost of Hospital Admissions.
9. This assumes, strictly speaking, that the slope of the Ostro relative risk function in Table 2.3 is approximately linear over the relevant range of ambient concentrations.
10. That is, 29.5 = (5384/365) * 2

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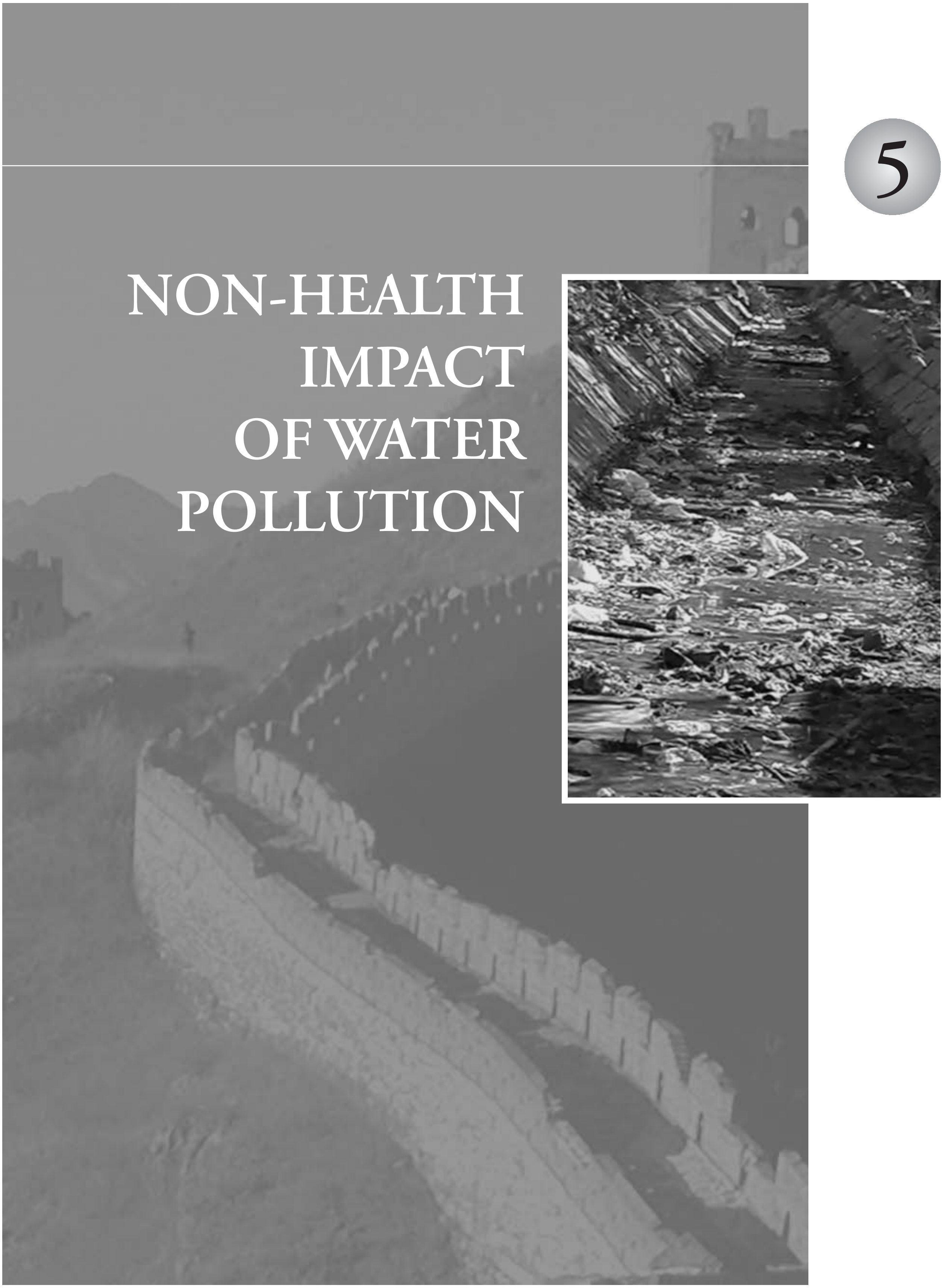
APPENDIX A.1 Average Life Years Lost due to Air Pollution

Age Groups	Remaining Life Expectancy	RD		CVD		CEVD	
		Deaths	Lost Life Years × Deaths	Deaths	Lost Life Years × Deaths	Deaths	Lost Life Years × Deaths
0–	78.79	1680.41	132393.79	266.69	21011.27	89.25	7031.41
1–4	78.51	518.07	40675.18	130.56	10250.80	21.93	1722.13
5–9	74.71	237.68	17756.99	68.13	5089.58	9.08	678.61
10–14	69.83	138.12	9644.70	138.12	9644.70	30.47	2127.51
15–19	64.92	195.99	12723.77	229.12	14874.26	99.38	6451.49
20–24	60.01	253.55	15215.41	548.45	32911.59	212.21	12734.64
25–29	55.15	480.56	26502.25	964.25	53176.60	436.88	24092.96
30–34	50.31	987.78	49696.39	2019.46	101601.51	1153.98	58058.01
35–39	45.50	1274.53	57991.06	3279.82	149232.08	2359.17	107342.37
40–44	40.73	1797.80	73231.00	4369.61	177990.75	4169.36	169833.60
45–49	36.07	3519.17	126930.67	7359.60	265448.36	8821.92	318191.93
50–54	31.49	4903.51	154393.74	8674.55	273130.10	10787.44	339657.12
55–59	27.06	6598.43	178537.66	9180.92	248413.60	12547.36	339501.39
60–64	22.86	14205.99	324757.07	16643.19	380473.08	21818.92	498793.14
65–69	19.01	25778.30	489933.95	29385.85	558497.67	36225.79	688495.36
70–74	15.64	41228.53	644924.21	39312.80	614957.14	47827.45	748148.97
75–79	12.96	46403.88	601328.17	40830.47	529104.65	48111.39	623455.11
80–85	11.07	41399.67	458171.04	34687.21	383884.08	36150.15	400074.47
85–	10.72	36785.43	394314.94	31578.30	338497.99	25073.53	268771.30
Total		228387.42	3809121.98	229667.09	4168189.83	255945.66	4615161.51
Average lost statistical years			16.68		18.15		18.03

Note: Deaths are the product of the population in the survey report of the national 5th population census and the disease-specific death rates in the Health Statistical Yearbook.
RD = Respiratory disease; CVD = Cardiovascular disease; CEVD = Cerebrovascular disease



5

NON-HEALTH IMPACT OF WATER POLLUTION

5.1 Water Scarcity and Pollution

Water scarcity is most prevalent in northern China. High pollution in this region exacerbates water scarcity. Polluted water is held back from supply and becomes a source of water scarcity. However, some water is allowed in the supply despite being too polluted; in such a case, pollution becomes a consequence of water scarcity. Groundwater depletion is a partly overlapping consequence of water scarcity that also creates a major environmental problem in China. We found that between 2000 and 2003 polluted water supply constituted about 47 billion cubic meters of water while polluted water held back from supply constituted about 25 billion cubic meters. Groundwater depletion constituted about 24 billion cubic meters. The economic cost of the pollution-related sources of water scarcity is estimated to be 147 billion RMB yuan, with a 95 percent confidence interval relative to uncertainty in valuation of 95 and 199 billion RMB. The cost of groundwater depletion comes at a further 92 billion RMB.

Water scarcity is predominantly an issue in northern China. While most of China's water resources are in the south, the greatest need for these resources is in the northern and eastern part of the country, where most of the people live. The four northern river basins contain less than 20 percent of national water resources, but account for two-thirds of the farmland and 45 percent of GDP. By contrast, the southwestern areas contain slightly more water resources (21.3 percent), but account for only 8.3 percent of GDP.[1]

The concentration of people and economic activity leads to water scarcity. Water scarcity has a number of different definitions. The United Nations Environment Programme (UNEP) defines it as a state in which "the amount of water withdrawn from lakes, rivers or groundwater is so great that water supplies are no longer adequate to satisfy all human or ecosystem requirements, bringing about increased competition among potential demands." (http://freshwater.unep.net.) The statement invites an interpretation in which there is a deficit of water: water withdrawn is larger than supply, which is no longer adequate.

An economist, on the other hand, would routinely define water scarcity as a situation in which demand for water, or water withdrawn, exceeds supply at a price of zero. This means that the available water is not sufficient for everybody to meet their needs at no financial cost. The economist's definition is echoed in part of the UNEP definition regarding competition among potential demands. In the absence of sufficient quantities, there must be competition between demands and an associated opportunity cost as reflected by the price of water.

In this chapter, we define water scarcity as a state in which available water resources per capita fall (far) below sustainable levels. Under such circumstances, there is a competition among potential demands and not all human and ecosystem requirements are met, as suggested by the UNEP definition. There is also a real risk that water supplies are no longer adequate to meet demand, and there is insufficient capacity to satisfy everyone's needs at a price of zero, as suggested by the economist's definition.

A combination of historical and contemporary trends can explain water scarcity in China's north and east. For various historical reasons, people have settled and prospered in the northern and eastern parts of the country, despite the low water resources, so contemporary China has inherited relatively high population density in these regions. Recent population growth and high economic growth have further increased the demand for water, while pollution of water basins as well as technical deficiencies in the water supply facilties, such as leaky pipes and canal, have reduced available water resources.

In this chapter, we are focusing on pollution as a cause of water scarcity. We attempt to estimate the economic costs that arise from the inability to make productive use of polluted water. In addition, we seek to identify environmentally unsustainable responses to water scarcity. In China, about 10 percent of the current water supply is too polluted to be usable. Pollution, therefore, increases the volume of water that is held back from use, but some of this water is supplied despite pollution, which also has a high economic cost. Industry and agriculture are the largest consumers of polluted water, despite fairly lax standards for what passes as acceptable water. We also attempted to estimate the environmental cost of consumption of such highly polluted water. In addition to pollution in the sense of not meeting the standards, there is also an issue of water that passes as acceptable but is fairly dirty. We do not attempt to estimate the environmental cost of that water.

Water scarcity in China also leads to depletion of groundwater resources. Depletion of groundwater resources, particularly deep aquifers, is another environmentally unsustainable response and adds to China's environmental costs. In some areas of China, the groundwater table has fallen more than 50 meters since 1960, and it continues to fall two meters annually.

Groundwater depletion is, to some extent, linked to the pollution problem. In several provinces, like in the lower reaches of the Yangtze, polluted water is withheld at the expense of more groundwater depletion. Furthermore, groundwater itself is often polluted by both natural and anthropogenic sources. An investigation of drinking water in 118 cities carried out by the Ministry of Water Resources (MWR) found that groundwater was polluted to varying degrees in 97 percent of the cities. Figure 4.6 shows that the two main groundwater pollutants are arsenic and fluoride.

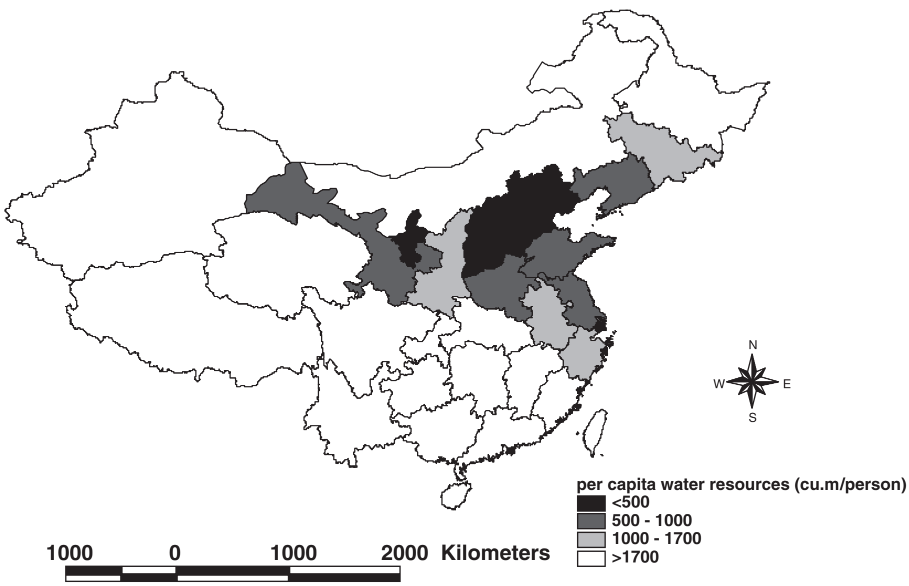
We sought to estimate the cost of groundwater pollution as a way of estimating pollution as a source of water scarcity. We also estimated total groundwater depletion, because it is a serious environmental problem and the method for separating out pollution related to groundwater depletion has some uncertainties. The cost of total groundwater depletion is not added to the total environmental cost estimate, but is available as stand-alone information.

WATER RESOURCES IN CHINA

To understand the nature of water scarcity in China, it is useful to begin with a survey of available, rechargeable water resources. A common indicator is the "water crowding" index of population per million cubic meters per year. Here we use the inverse measure of cubic meters per person. Levels of 1,000–1,700 cubic meters per person indicate water stress, and less than 1,000 cubic meters per person indicates extreme water scarcity (Vörösmarty et al. 2006; Falkenmark 1997). Figure 5.1, with data from NBS (2004), shows that in six provinces in China, per capita water resources fall below 500 cubic meters.[2] In a further five provinces, water resources falls below 1,000 cubic meters, meaning that one-third of China's provinces qualify for extreme and more than extreme water scarcity.[3] Available resources depend on precipitation. Data for water resources are from 2003, which was an average year in terms of water resources nationwide.[4]

The figure shows that per capita water resources are lowest in the Huang-Huai-Hai river

FIGURE 5.1 Per Capita Water Resources in China in 2003



basins, especially the lower reaches. The lower reaches of the Huang River, also called the Yellow River, used to dry up until the construction of the Xiaolangdi Dam (Berkoff 2003). The middle reaches of the Huang River have also experienced dry spells (Zhu 2006). The middle and lower reaches of some tributaries of the Hai River tend to be dry all year round.

On the other hand, several provinces in southwestern China have abundant water resources, including Yunnan, Qinghai, and Tibet. The Yangtze River (Changjiang), Pearl River, and the rivers in the east and south together have 80 percent of the water resources of China (see Table 5.1) (MWR 2005a). The average per capita water resources for all of China was 2,131 cubic meters in 2003.

The scarcity of water resources in the Huang-Huai-Hai is particularly pressing in dry years. The Hai and Huai flows fall to 70 percent of average in one year in four and to 50 percent one year in twenty (Berkoff 2003). Dry years tend to come in succession, accentuating the problem.

POLLUTION AS A SOURCE OF WATER SCARCITY

As discussed in chapter 3, the most polluted water basins in China are located in the northern and eastern parts of the country in the same regions that have low water resources per capita. Since water is polluted, less is available for consumption in households, industries and agriculture, which further exacerbates the serious water scarcity situation.

The decrease in water consumption below levels needed by households, industries and agriculture, called repressed demand, is one possible impact of low water resource availability. Another impact may be increased groundwater depletion, which may happen if water authorities consider it a priority to maintain water supply. Increased reliance on groundwater is also the decentralized response of farmers, who may dig wells when they are not allowed surface water for irrigation. Anecdotal evidence and the expert opinion of MWR suggest that groundwater depletion is in

TABLE 5.1 The Quantity of Water Resources in China, Average 1956–2000

Water Resource Region (I class)	Precipitation (billion m³)	Surface Water (billion m³)	Shallow Rechargeable Groundwater (billion m³)	Non-Overlapping Quantity (billion m³)	Total (billion m³)
Songhuajiang River	471.9	129.6	47.8	19.6	149.2
Liaohe River	171.3	40.8	20.3	9.0	49.8
Haihe River	171.2	21.6	23.5	15.4	37.0
Yellow River	355.5	59.4	37.8	11.2	70.7
Huaihe River	276.7	67.7	39.7	23.9	91.6
Changjiang River	1937.0	985.7	249.2	10.2	996.0
Southeast rivers	437.2	265.4	66.5	2.7	268.1
Pearls River	897.2	472.3	116.3	1.4	473.7
Southwest rivers	918.6	577.5	144.0	0.0	577.5
Northwest rivers	542.1	117.4	77.0	10.2	127.6
Total	6178.7	2737.4	822.1	103.6	2841.2

Source: Ministry of Water Resources.

fact a common alternative to polluted surface water.

There are no readily available statistics on polluted water that is held back from water supply in China. To estimate the amount of water held back, we rely on the assumption that groundwater depletion and repressed demand are the two responses to holding back water. See Box 5.1 for details about the method.

Table 5.2 presents the estimate of non-supplied polluted water by province. Hebei Province and Shandong Province are estimated to have the largest volume of non-supplied polluted water. Ningxia Autonomous Region and Shanghai both have zero non-supplied polluted water. The reasons are rather different, however. In Ningxia, the situation is so tight that all available water resources, including all available polluted resources, are used in supply. In Shanghai, the supply of water is greater than demand for water, and there is no recorded depletion of groundwater. Shanghai's situation should probably be viewed in the context of neighboring Jiangsu Province, which has the third highest non-supplied polluted water volume in the country. Jiangsu Province is the home of the lower reaches of the Yangtze, mentioned in the introduction as an area where groundwater depletion substitutes for polluted water.

IMPACT OF WATER SCARCITY: WATER POLLUTION IN SUPPLY

Water scarcity has led China to make use of excessive amounts of polluted water in its water supply. Polluted water is supplied to households, industry and, in particular, agriculture. Water for households and industry is in most cases treated before consumption. Impacts of household consumption of polluted water are discussed in chapter 3. Impacts of wastewater irrigation are discussed in this chapter. Impacts on industries include lower product quality and production stoppages. For instance, a report by Chang, Seip, and Vennemo (2001) from a Chongqing silk production plant found that the raw silk became yellow when polluted water was used, and its quality fell from 5A to 3A or 2A. Another silk production plant and a fertilizer plant were forced to stop production.

This chapter provides a comprehensive picture of the extent of polluted water in supply.

BOX 5.1 Constructing an Estimate of Polluted Water Held Back from Supply

We distinguish between water-scarce water basins on the one hand, and water-abundant water basins on the other. Water-scarce areas not only have little water resources per capita—the indicator we emphasized above—but also have high consumption rates relative to their water resources. In the literature, a consumption rate of 20–40 percent is considered medium to high (World Meteorological Organisation 1997; Vörösmarty et al. 2006). We define a water-scarce basin in China as one in which the consumption rate is above 40 percent. We made separate calculations for each of the 73 water basins and provinces distinguished by MWR.

In water-scarce basins, we assume that the quantity of polluted water that is held back equals *either* the quantity of depleted groundwater *or* a measure of repressed demand, whichever is largest, times the share of polluted water in the water resource. This share is defined as the share of water of quality IV and worse. That is, all water that is unsuitable for bodily contact is considered polluted. To measure the share, we take the weighted sum of river sections in which measured water is polluted divided by the total weighted sum. The weights are the lengths of the river sections. In effect then, the shares give polluted water in the rivers and basins measured by length. In symbols:

(1) $$PW = \alpha \max(G, RD)$$

In this equation, *PW* represents polluted water that is withheld, *G* is groundwater depletion, *RD* is repressed demand, and α is the share of polluted water in the resource. The idea here is that the more pollution there is in the resource, the greater is the share of groundwater depletion or repressed demand that can be attributed to non-supplied polluted water. If the share is zero, then none of the (presumably low) amount of water depletion or repressed demand should be attributed to pollution. If the share is one, then all water depletion or repressed demand should be attributed to pollution.

Repressed demand is calculated as the difference between notional demand, which is a planning indicator of MWR, and sustainable supply; that is, current supply excluding groundwater depletion and supply of polluted water.

For example, in Shanxi Province repressed demand is estimated to 0.76 billion cubic meters. That is larger than the volume of groundwater depletion, estimated at 0.54 billion cubic meters. In Shanxi Province, the share of polluted water is 71 percent. Polluted non-supplied water is estimated to be 71 percent of 0.76 billion cubic meters, or 0.54 billion cubic meters. By coincidence, that number corresponds to the volume of groundwater depletion. That is not the case in all provinces.

There is one exception to equation (1). It is possible to estimate how much water is withheld in a water basin *in total.* If that volume is lower than the estimate coming out of equation (1), there is obviously a problem with the estimate from equation (1). The volume of polluted water that is held back cannot be higher than the volume of all water that is held back. To eliminate this possibility, we assume that when the total volume is lower than the largest of groundwater depletion and repressed demand, then polluted withheld water equals total available water times the polluted share. In symbols

(2) $$PW = \alpha TW, \; TW \leq \max(G, RD)$$

It is a complicated matter to estimate *TW,* available total water. First, we calculate the total amount of polluted water in the resource, using estimates of the total surface water resource (the basis for the 40 percent or more consumption rates) and the polluted share. From this volume, we subtract the amount of polluted water that is supplied. In some basins, there is also some residual clean water not supplied, which is added to the total resource. The net result of these operations is the estimate of available total water TW in a water basin.

For example, in Tianjin the available total water is estimated to be 0.212 billion cubic meters. That is slightly lower than either repressed demand or groundwater depletion, which both amount to about 0.215 billion cubic meters. The estimate of 0.212 in available total water equals

(*continued*)

BOX 5.1 Constructing an Estimate of Polluted Water Held Back from Supply (*Continued*)

the total surface water resource of 1.1 billion cubic meters times a polluted share of 86 percent. From this number is subtracted the amount of polluted water that is supplied, which is as high as 0.744 billion cubic meters (see Table 5.). There is no surplus clean water in Tianjin.

With 0.212 as the estimate of total available water in Tianjin, we use equation (2) and estimate polluted withheld water to be 86 percent of 0.212 billion cubic meters, which is 0.18 billion cubic meters.

Equations (1) and (2) apply to river basins in which consumption is 40 percent or more of the rechargeable resource. When consumption is lower than 40 percent, there is no resource-oriented reason for groundwater depletion. The expert judgment of MWR and SEPA is that in river basins of less than 40 percent consumption, the reason for groundwater depletion is that available resources are too polluted for use. There is normally no repressed demand in these river basins. Accordingly, we assume that in these basins non-supplied polluted water equals water depletion. In symbols

$$(3) \quad PW = G, \; consumption \leq 40\%$$

Fifteen southern provinces—including Jiangsu, Sichuan, and Guangdong—are in a situation where consumption is lower than 40 percent.

The procedure to estimate non-supplied polluted water relies on a number of untested assumptions. Yet SEPA and MWR consider that it gives a rough indication of non-supplied polluted water in China.

By polluted water in supply, we refer to water that exceeds the water quality standard relevant for the purpose. Polluted water for households refers to water worse than class III supplied (after treatment) to households; polluted water for industrial purposes refers to water worse than class IV supplied (after treatment) to industry; and polluted water for agricultural purposes refers to water worse than class V supplied to agriculture.

Figure 5.2 shows the volume of water supply (in millions of cubic meters) that does not meet supply standards for each province. The map was produced using MWR survey data from 2000 to 2003. The multiyear coverage allows us to account for annual variations due to rainfall and other factors.

The map shows significant correlation with the map of water resources. For instance, Ningxia and Shanghai are the provinces in the country with the lowest per capita water resources in 2003. Ningxia, Jiangsu, and Heilongjiang are the provinces with the largest supply of polluted water (9.5 billion cubic meters in Jiangsu, 4.0 in Heilongjiang and Ningxia). Note that the correlation is specified between a per capita measure and an aggregate measure. The amount of pollution in Ningxia is particularly remarkable, since it has a very small population. Jiangsu is situated in the lower reaches of the Yangtze River. On its border is the final destination of the Huai River, which never reaches the sea. Ningxia is in the upper reaches of the Huang (Yellow) River.

While the correlation is striking, there are also differences with the map of water resources per capita. Supplies of polluted water in the Hebei-Beijing-Tianjin area are large, but relative to other provinces the situation is better than it is in terms of water resources. Heilongjiang Province, by contrast, has a serious problem with the supply of polluted water compared to its water resources. Altogether, close to 50 billion cubic meters of water that did not meet the pollution standard were supplied annually during the 2000–03 period. This figure is close to 10 percent of average national water consumption in the period, which was 566 billion cubic meters.

TABLE 5.2 Non-Supplied Polluted Water by Province

Province	Polluted, Non-Supplied Water (million m³)
Beijing	138.1
Tianjin	184.2
Hebei	3,618.7
Shanxi	540.6
Inner Mongolia	2,498.1
Liaoning	613.7
Jilin	317.4
Heilongjiang	970.8
Shanghai	0.0
Jiangsu	2,165.8
Zhejiang	496.4
Anhui	1,647.7
Fujian	349.4
Jiangxi	319.3
Shandong	2,716.9
Henan	1,991.4
Hubei	367.1
Hunan	353.5
Guangdong	1,083.9
Guangxi	572.5
Hainan	135.9
Chongqing	311.1
Sichuan	697.1
Guizhou	361.6
Yunnan	1,058.1
Tibet	175.6
Shaanxi	520.5
Gansu	400.2
Qinghai	32.0
Ningxia	0.0
Xinjiang	124.9
Total	24,762.4

Source: Authors Calculation.
Note: Polluted water is water of class IV or worse.

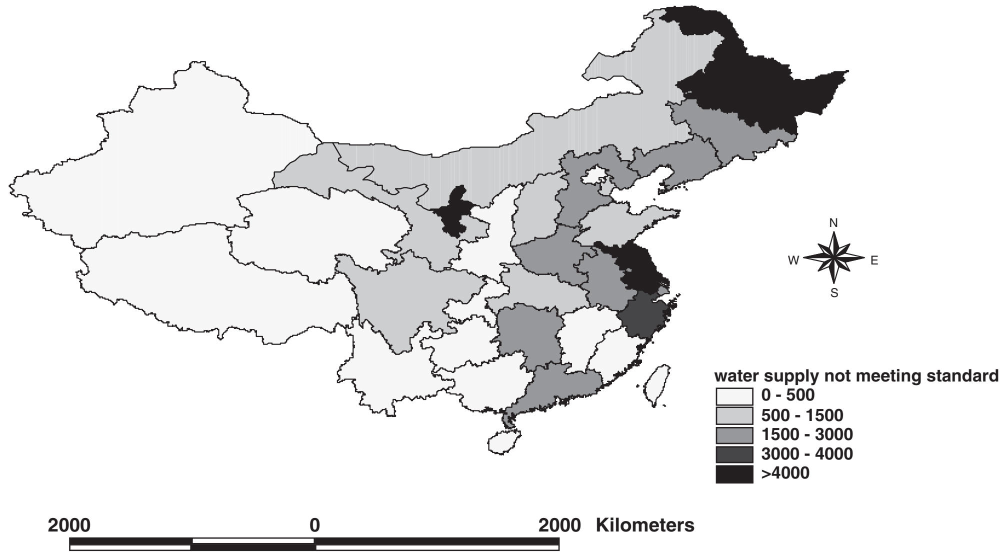
A breakdown of polluted water supply by consumption sector shows agriculture receiving two-thirds of the water, and industry receiving 20 percent. The ratios differ by province. In the two high consumption provinces of Jiangsu and Ningxia autonomous region, agricultural consumption constitutes 62 percent and 98 percent (respectively) of all consumption. In Ningxia, it has been reported that a local tradition of applying sediments as fertilizer encourages excessive use of irrigation water (Yang, Zhang, and Zehnder 2003). Pollution of irrigation water is related to about 62 billion cubic meters of sewage in China in 2000 (MWR, 2000), of which only 24 percent was treated up to standard.

IMPACT OF WATER SCARCITY: DEPLETION OF GROUNDWATER

Water scarcity has forced China to rely increasingly on groundwater, which has led to depletion of groundwater reservoirs. It is useful to distinguish between rechargeable groundwater in the shallow freshwater aquifer (phreatic water), and non-rechargeable groundwater in the deep freshwater aquifer (confined water). Shallow groundwater recharges from, and/or discharges to, precipitation and surface water flows; for example, compare the columns *shallow rechargeable groundwater* and *non-overlapping quantity* in Table 5.1. Depletion of shallow groundwater occurs when consumption exceeds sustainable levels. Deep groundwater recharges/discharges extremely slowly. Replenishment rates can be in the order of thousands of years. Depleting deep groundwater is similar to mining a nonrenewable resource.

Depletion of groundwater may have serious consequences for the environment. One is salinity intrusion, as declining groundwater resources are substituted by brackish water that often lies between the shallow and deep groundwater tables. Salinity intrusion is also caused by seawater intruding from the outside. Land subsidence following compaction of the geological formation containing groundwater (so-called aquitard) is another unfortunate consequence of groundwater depletion. In China, salinity intrusion is a chronic problem, such as in the Hai River Basin (Zhu 2006). In some locations, intrusion of brackish water has been monitored at a rate of 0.5–2 meters per year for the past 20 years (Foster et al. 2004). In turn, salinity intrusion poses problems for the waterworks and for human

FIGURE 5.2 Polluted Water in Supply in China



Source: Ministry of Water Resources

and agricultural use. A recent episode of this kind occurred in the Pearl River delta in the south. MWR (2005) reports that in 2003–04, following a period of 30 percent lower precipitation than normal, the Pearl River delta suffered a case of severe salinity intrusion. The salinity intrusion restricted operation of the waterworks of Pearl River and Macao for 170 continuous days. More than 5 million people, as well as industries and agriculture, were affected to varying degrees.

Both for natural and man-made reasons, the quality of groundwater often is poor. An investigation of drinking water in 118 cities carried out by MWR found that groundwater was polluted to varying degrees in 97 percent of the cities. In 64 percent of cities, groundwater was seriously polluted. Data from MWR indicates that in 30 percent of the area supplied by groundwater, people should not use the water for drinking purposes.[5] This area contains a disproportionate number of cities. In a further 30 percent of the area, groundwater needs to undergo water treatment.[6]

Besides the environmental aspects, groundwater depletion carries an economic cost. As the groundwater table falls, the cost of pumping it becomes high, especially for agricultural purposes.

Figure 5.3 shows groundwater depletion by province in the period 2000–03 and refers to the survey of MWR mentioned above. The figure refers to depletion of shallow and deep groundwater in total.

The figure shows that depletion of groundwater extends from the Huang-Huai-Hai plain to almost every province in the north, including heavy depletion in Inner Mongolia, and substantial depletion in Xinjiang. It reinforces the message that the north and east have the most serious problems of water scarcity and are the main source of water depletion. Particular problems are evident in Hebei Province and surrounding provinces, most of which belong to the Huang-Huai-Haiplain.

In Hebei, 6 billion cubic meters of groundwater were depleted annually in 2000–03. Zhu (2006) comments that part of the aquifer in

TABLE 5.3 Supply of Surface Water that Does Not Meet Pollution Standards ($10^6 m^3$)

Province	Urban Domestic Use	Rural Domestic Use	Industrial Use	Large Irrigation Use	Other Agricultural Use	Surface Total
Beijing	0	0	0	100	0	100
Tianjin	0	0	13	731	0	744
Hebei	0	0	14	910	1159	2083
Shanxi	30	69	105	342	612	1158
Inner Mongolia	13	0	67	468	194	742
Liaoning	41	0	381	2049	233	2703
Jilin	309	0	55	190	1084	1639
Hailongjiang	289	0	1190	354	2211	4045
Shanghai	1184	74	1018	0	646	2922
Jiangsu	707	609	2281	280	5629	9508
Zhejiang	379	215	1253	805	1011	3663
Anhui	133	254	134	0	1919	2440
Fujian	1	12	0	0	0	13
Jiagxi	4	0	141	0	67	211
Shandong	52	2	14	15	503	586
Henan	226	5	408	639	1396	2674
Hubei	101	5	334	10	54	504
Hunan	305	270	648	0	627	1850
Guangdong	1138	50	1276	0	100	2563
Guangxi	19	53	124	0	88	284
Hainan	0	0	0	0	0	0
Chongqing	69	69	103	0	36	277
Sichuang	247	126	150	0	73	596
Guizhou	1	0	50	0	1	52
Yunnan	0	0	0	0	0	0
Tibet	0	0	0	0	0	0
Shaanxi	4	3	0	290	0	297
Gansu	252	46	167	0	531	997
Qinghai	12	1	14	0	189	216
Ningxia	0	2	135	2162	1655	3953
Xinjiang	0	0	0	32	0	32
Total	5516	1866	10073	9348	20018	46821

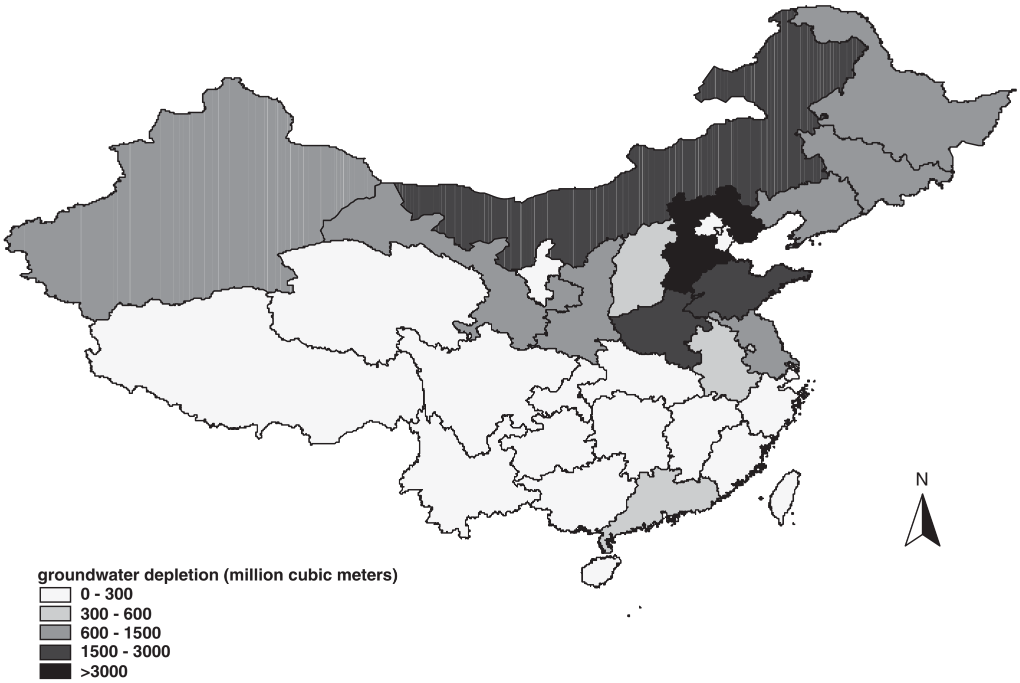
Source: Authors calculation.

Hebei and Beijing are nearly dried up, and in other parts the groundwater table is sinking 3–5 meters annually. In most of the Huang-Huai-Hai plain, the groundwater table has dropped. The drop is 2–3 meters in some areas and as much as 10–30 meters in others. Foster et al. (2004) state that in rural areas of the Huang-Huai-Hai plain "an average value for deep aquifer groundwater-level decline of more than 3 m/year during the period 1970–80 has now reduced to 2 m/year." Compounded over 35 years, this implies a deep-aquifer depletion of more than 50 meters.

Table 5.4 indicates amounts of annual groundwater depletion in 2000–03 between provinces, as well as its distribution between urban and rural households, industry, and agriculture. Groundwater depletion totaled 24 billion cubic meters.

As can be seen from the table, a common use of groundwater is for irrigation of agriculture. In fact, 74 percent of all groundwater depletion is for agricultural purposes. In many areas irrigated by

FIGURE 5.3 Groundwater Depletion by Province (million cubic meters)



Source: Ministry of Water Resources

groundwater, each pump serves only a small number of farmers (Yang, Zhang, and Zehnder 2003). It is a sign both of the strain on water resources and of the spread of groundwater depletion that in 1997 alone, 221,000 wells were drilled on the Huang-Huai-Hai plain, while 100,000 wells were deserted. In Beijing and Tianjin, the numbers of newly drilled wells were outstripped by those deserted (Yang and Zehnder 2001).

THE ENVIRONMENTAL COST OF WATER SCARCITY

We have found that approximately 25 billion cubic meters of polluted water in China is held back from water consumption, contributing to problems of repressed demand and groundwater depletion. As much as 47 billion cubic meters of water that does not meet quality standards is nevertheless supplied to households, industry, and agriculture. A further 24 billion cubic meters of water beyond rechargeable quantities is extracted from wells and creates groundwater depletion. Although there are some overlaps, close to 100 billion cubic meters of water in China is affected by pollution and other environmental stress. By comparison, the total supply of water in China is approximately 550 billion cubic meters (NBS 2006).

Most experts agree that water scarcity substantially restricts economic development. According to Zhu (2006), water scarcity becomes “an obstacle for the enhancement of people’s living standard as well as construction and development of big water-consuming industrial enterprises.” Furthermore, water scarcity “restricts agricultural development in North China and threatens food safety."

Another impact of water scarcity is to increase the frequency and force of droughts. Droughts lead to economic loss and human strain. In 2004, for example, 218 million *mu* of cropland were damaged by drought, causing 19.9 million tons of damage to grain production and an economic loss of 24.7 billion yuan. As a result of this drought, 23.4 million people were left temporarily without drinking water supply (MWR, 2004). According to MWR, more severe droughts had preceded the one in 2004.

Taking arguments like these forward, several authors have tried to systematically assess the value of water in China. Box 5.2 describes some of these efforts.

Our conclusion from going through the evidence is that generally 1–5 RMB yuan per cubic meter seems a fair value for water in China. To be more concrete, one should consider what would have happened to the water if the environmental problem was not there. In terms of polluted water held back from supply, the value of marginal water seems a reasonable indicator. To value this water, we use the paper of He and Chen (2005), which besides producing estimates that are a priori reasonable compared with other sources, is a recent and comprehensive attempt. Given the rapid economic growth in China, value added per unit of water increases considerably year by year. Recall, for instance, the increase over time in agricultural output per unit of water. It is, therefore, particularly important to use a recent estimate.[7]

To value polluted water that is included in supply, we could in principle also use the estimates of He and Chen (2005). However, polluted water used for irrigation purposes in agriculture, particularly in so-called wastewater irrigation zones, is discussed later in this chapter. To avoid overlap, we focus on polluted supply for households and for industry, which amount together to around 17 billion cubic meters. This water needs treatment before consumption, and the cost of treatment is part of the environmental cost. Other costs related to polluted supply for households, including health costs, are discussed in chapter 4. Costs to industry in addition to treatment, such as halts in production, are not included.

Groundwater depletion can ideally be valued by its environmental effects. As noted above,

TABLE 5.4 Depletion of Groundwater

Depletion of Groundwater ($10^6 m^3$)

	Quantity				
Province	**Urban Domestic**	**Rural Domestic**	**Industrial**	**Irrigation**	**Total**
Beijing	74	8	64	114	261
Tianjin	45	8	49	113	215
Hebei	316	331	909	4553	6109
Shanxi	37	35	108	363	543
Inner Mongolia	75	87	159	2406	2728
Liaoning	127	51	280	823	1280
Jilin	37	28	121	419	605
Heilongjiang	67	39	305	1078	1489
Shanghai	0	0	0	0	0
Jiangsu	64	44	364	703	1175
Zhejiang	11	6	32	66	116
Anhui	21	47	144	318	530
Fujian	0	0	0	1	2
Jiangxi	0	0	0	0	0
Shandong	138	153	480	1894	2665
Henan	139	242	434	1655	2469
Hubei	7	4	31	58	101
Hunan	2	3	9	32	46
Guangdong	47	24	121	268	459
Guangxi	5	10	18	111	144
Hainan	7	8	5	102	122
Chongqing	0	0	0	0	0
Sichuang	1	1	2	5	9
Guizhou	0	0	1	2	3
Yunnan	3	4	6	49	63
Tibet	0	0	0	0	0
Shanxi	62	57	166	686	971
Gansu	17	19	86	486	608
Qinghai	0	0	0	3	4
Ningxia	2	1	8	146	158
Xinjiang	15	14	26	1309	1363
Sum	1320	1224	3930	17763	24236

Source: Authors Calculation.

BOX 5.2 Efforts to Value Water in China Relevant for Water Scarcity

Efforts to estimate the value of water relevant for water scarcity usually follow one of two approaches. One approach is to *estimate the value of marginal water.* Estimating the value of marginal water means finding the economic value added if a little extra water were available in the economy—or the value deducted if a little less water were available. Using this approach and a so-called computable general equilibrium model, He and Chen (2005) find that a cubic meter of water adds between 2.1 and 5.2 RMB yuan of value. The value differs between water basins. The highest values are obtained in the Huang-Huai-Hai basins, which is reasonable, since these are the most water scarce basins. By comparison, an earlier paper by Liu and Chen (2003) uses a linear programming model and 1999 data to find that water for industrial purposes adds between 0.12 and 9.07 RMB yuan per cubic meter between provinces. The highest value is obtained for Ningxia Province.

Several authors use the marginal value method in an informal way. Foster et al. (2004) argue that agricultural irrigation is the marginal use of water on the Huang-Huai-Hai plain. It is the sector that accommodates additional water (rainfall) and the sector that suffers during droughts. It is also the sector that would notice the shortfall if groundwater depletion was disallowed. The main crop affected by water scarcity is winter wheat. Liu and He (1996) report that on the Huang-Huai-Hai plain, 1.2 kg of wheat is grown per cubic meter of water (cited in Yang and Zehnder 2001). Jia and Liu (2000) estimate that in Shaanxi Province the figure is 1.3 kg, and increasing. (Their estimate for 1981 is 0.6 kg). Other authors, including Foster et al., use a somewhat lower number. With a national wholesale price of wheat of 1.15 RMB yuan/kg (see chapter 6 on wastewater irrigation), the implied value of water is currently approximately one RMB yuan per cubic meter, but as mentioned earlier, this estimate depends on irrigation of winter wheat being the marginal use of water.

The marginal value of water can also be indicated by its price. It is likely that water is purchased to the extent that the value of water to the consumer is at least as high as the price; that is, the marginal value of water equals the price. With repressed demand, however, the marginal value is probably higher than the price. Increasing water prices is, therefore, included in MWR's strategy for a "water-saving society." Currently the average price of water in China's 36 major large and medium-sized cities is 2.1 RMB yuan per cubic meter (MWR 2006). This price refers to urban and domestic use. Prices have recently increased 10 percent annually, and will have to increase even more to make a serious impact on water consumption. Still, 2.1 RMB yuan is an estimate of the value of water from the side of the price. Of course, in some circumstances water has a much higher price—up to 2 RMB yuan per *liter* for bottled drinking water. However, in such cases the circumstance and packaging is part of the product.

Another main approach to valuation of water is e*stimating the cost of current mitigation measures.* While in some cases estimating environmental cost via the cost of mitigation contains an element of circular reasoning, it is in other cases a useful measure of the political willingness to pay, or valuation of water. Other things being equal, it is the approved mitigation measures with the highest cost that come the closest to expressing a political valuation for water. A main element in China's strategy to end groundwater depletion is the South-North Water Diversion Project, which will transfer water from the Yangtze River to the Huang-Huai-Hai basin. The project will move up to 45 billion cubic meters annually to the basin. That number equals half of current water scarcity as estimated by this chapter. Water demand in China is likely to be significantly higher upon completion of the project (around 2050) than it is now. Still the project indicates a Chinese willingness to reduce and perhaps end pollution-related problems of water scarcity.

The investment cost of the South-North Water Diversion Project is tentatively set at RMB yuan 486 billion, but after two years of investment it faced a 20 percent cost overrun (China Daily 2004). Ignoring cost overruns, Berkoff (2003) finds that the implied annual value of water is 0.7–0.9 RMB yuan per cubic meter. Allowing 20 percent higher investment cost increases these numbers to 0.9–1.1; adding in 0.06–0.38 RMB yuan annual operation and maintenance cost per cubic meter increases them to 1.2–1.3 RMB yuan per cubic meter. These estimates assume a 12 percent rate of return on the investment, which despite high economic growth in China and associated high return to capital, still could be on the high side. A high rate of return in the investment implies a high value of the future benefit stream, which is a high value of the diverted water.

(*continued*)

BOX 5.2 Efforts to Value Water in China Relevant for Water Scarcity (*Continued*)

Other mitigation options that have been discussed in the literature are related to agriculture in particular. They include agricultural crop changes (Yang and Zehnder 2001), efficient water irrigation techniques (Foster et al. 2004) and desalination (Zhou and Tol 2003), and import of agricultural produce. Import of agricultural produce is sometimes referred to as import of "virtual water," since it is an indirect way of transporting water to an area (Allen 1993).

When polluted water is supplied, it requires treatment. A survey by CAEP (2006) of about 1,000 enterprises in ten provinces has estimated that the treatment cost for domestic purposes is about 2.6 and that for industry is 4.6 RMB yuan per cubic meter. The estimates are preliminary and will need more in-depth analysis.

groundwater depletion may lead to salination and to compaction of land. The main cost, however, may relate to groundwater's existence value and the speed of resource exhaustion; that is, water that has been accumulating underground over thousands of years is being spent by only a few generations. This is a cost that is relevant and of concern not only for the Chinese, but the international community as well.

The environmental cost of groundwater depletion can be estimated directly, if one focuses on the salinity issue, but ignoring the importance of existence value may lead to significant underestimation of the real cost. Therefore, we consider the marginal economic value of groundwater on the assumption that the environmental cost of groundwater is as least as high as its economic value.

To say that the cost of groundwater depletion is higher than its economic value is an untested assumption that relies on water depletion being a rational choice of society. Had it been a rational choice, the implicit value of untapped groundwater would have been lower than the economic value. While untested, there is general agreement among experts that the present depletion of groundwater is the consequence of decentralized decisions without appropriate incentives for conservation. For instance, individual farmers in groundwater irrigated areas usually do not pay for the water itself. Farmers only pay for power and equipment (Yang, Zhang, and Zehnder 2003). When farmers are not informed by the market of any cost of groundwater depletion, it goes without saying that depletion ends up higher than a social planner would have desired. In terms of institutional distortions, MWR has recently emphasised the need to develop a water resources management system on the basis of the theory of water rights and water markets (MWR 2005), and the World Bank has long advocated water management reform in China. There are thus reasons to assume that present groundwater depletion is not the outcome of rational choice.

Using the values that we have described, Table 5.5 sets out the environmental cost of water scarcity by province. In the final estimate of environmental cost, we do not include the cost of groundwater depletion. One reason is that groundwater depletion is not wholly a pollution-related item. Another reason is that there is overlap between groundwater depletion and polluted water held back from supply. Including both items in the estimate of environmental cost would imply double-counting. Note also that the cost of wastewater irrigation is discussed in section 5.2, while the health cost of water pollution is discussed in chapter 3.

The environmental cost of water scarcity related to pollution amounts to 147 billion RMB. Among the provinces, Hebei and Jiangsu provinces have the largest environmental cost. Hebei is dominated by the Hai River basin. Jiangsu is split almost equally between the Huai and Yangtze. The cost of groundwater depletion amounts to 92 billion RMB.

TABLE 5.5 The Environmental Cost of Water Scarcity

Province	Cost of Polluted Water Held Back from Supply (1)	Cost of Polluted Water in Supply (2)	Environmental Cost—Sum of (1) and (2)	Cost of Groundwater Depletion (3)
Beijing	721	0	721	1,360
Tianjin	961	60	1 022	1,122
Hebei	18,790	63	18,853	31,744
Shanxi	2,547	565	3,112	2,595
Inner Mongolia	9,431	344	9,775	10,735
Liaoning	1,881	1,872	3,753	3,925
Jilin	971	1,072	2,044	1,853
Heilongjiang	2,971	6,277	9,248	4,556
Shanghai	0	7,850	7,850	0
Jiangsu	5,745	12,437	18,182	3,217
Zhejiang	1,081	6,806	7,886	270
Anhui	4,648	971	5,619	1,438
Fujian	785	4	789	4
Jiangxi	921	662	1,583	0
Shandong	9,477	202	9,679	9,060
Henan	7,384	2,489	9,874	8,372
Hubei	1,061	0	1,061	290
Hunan	1,020	3,807	4,827	133
Guangdong	2,863	8,921	11,784	1,213
Guangxi	1,515	625	2,140	381
Hainan	359	0	359	321
Chongqing	660	659	1,319	0
Sichuan	1,478	1,349	2,826	19
Guizhou	1,022	236	1,257	9
Yunnan	2,623	0	2,623	156
Tibet	556	0	556	0
Shanxi	2,196	11	2,207	3,988
Gansu	1,237	1,440	2,677	1,861
Qinghai	126	98	223	12
Ningxia	0	625	625	722
Xinjiang	400	0	400	4,360
Sum	85,429	61,258	146,687	92,356

Note: For polluted water held back from supply the marginal value approach is used, He and Chen (2005). For polluted water in supply the mitigation cost approach is used, that is treatment cost, CAEP (2006). For groundwater depletion He and Chen (2005) is again used. Groundwater depletion is only available by province. Water basin values of He and Chen are aggregated using supply per water basin in a province as weights.

UNCERTAINTIES AND SENSITIVITY

There are uncertainties both in the quantity and value aspect of the environmental cost of water scarcity. Quantities for polluted water in supply and groundwater depletion come from the MWR survey data from 2000–03, so there is significant uncertainty associated with these figures. With limited knowledge of the sampling method, we subjectively estimate the uncertainty to about ±20 percent. The quantity of polluted water held back from supply is estimated by the authors and the uncertainty of the calculation is estimated to about ±40% percent.

The values used also are uncertain. The average price used to value the cost of polluted water held back from supply is 3.45 yuan RMB per cubic meter (marginal production value). The average price used to value polluted water in supply is 3.93 yuan RMB per cubic meter (treatment cost). The average price used to value groundwater depletion is 3.81 (marginal production value, but different composition between provinces than polluted water held back from supply). While it is reassuring that the prices from different sources are similar, there are several sources of uncertainty. On the side of production value, the price depends on which sector and activity is marginal in the Chinese economy. If the marginal activity is agriculture, in particular winter wheat, the value of water may go as low as one RMB yuan per cubic meter. If the marginal value is industry, the value may go above six RMB yuan per cubic meter. A standard deviation of ±1 on the estimate of 3.45 (3.81) seems reasonable, and (1.45, 5.45) would then give a 95 percent confidence interval.

On the side of treatment cost, CAEP is currently reanalyzing the data. Based on preliminary analysis ±0.5 is a reasonable standard deviation, and thus (2.93, 5.93) is a 95 percent confidence interval. The uncertainty is larger on the side of industry treatment cost.

Treating the quantities as givens, we obtain a 95 percent confidence interval on the environmental cost of (95, 199) billion RMB yuan.

In addition to the quantified uncertainty, there are several omissions that contribute to making our estimate of environmental cost imprecise. For instance, the cost of treating water that is within quality limits but is polluted is omitted from our estimate.

5.2

Crop Loss Due to Wastewater Irrigation

A combination of water scarcity, growing demand for agricultural products, and readily available wastewater supply has contributed to continuous increase in wastewater irrigation in China for a number of decades. Wastewater-irrigated areas increased by a factor of 1.6 between 1982 and 1995. This report estimated that in 2003, wastewater irrigation areas totaled about 4.05 million hectares. It is estimated that the economic cost of wastewater irrigation on four major crops (wheat, corn, rice, and vegetables) in China is about 7 billion RMB annually. This cost estimate was arrived at by accounting for the impact of polluted water on crop quantity and quality, including fitness for consumption as well as the impact on the crops' nutrition quality. Economic losses due to ecosystem degradation and damage to human health were not included in this analysis, which means that the total economic cost associated with wastewater irrigation is most likely larger.

The output and quality of agricultural crop production is in many areas in China seriously affected by water pollution from wastewater irrigation. As noted in other chapters of this report, there is a serious shortage of water resources in China, especially in the north. To mitigate the problem, it is quite common to use wastewater or sewage for irrigation.

Although water pollution damage to farm crops is recognized as a common problem in China, little research is carried out to document the problem (ECON 2000). Two comprehensive investigations of effects on crops in areas irrigated with water from sewage pipes, industrial plants, and other wastewater sources have been conducted. The wastewater/sewage irrigation zone refers to farmland with an area over 20 hectares irrigated with water that either does not meet the government standards for water quality in farmland irrigation or for any other reason may lead to the death of aquatic species such as fish or shrimp (GB5084-92)[2]. The first survey (MoA 1984) was conducted over 20 years ago and the results are probably not a very good reflection of the present situation. A second survey in 1998 included two parts: (1) a general survey of national wastewater irrigation areas; and (2) some data on irrigation water quality, pollution conditions of the farmland, and crop quality in representative wastewater irrigated areas (MoA 2001). Although the second survey is also not very recent, in the absence of more up-to-date data, we decided to use it in the present evaluation.

In the second survey, water samples were collected for short time spans and not regularly throughout the entire growth periods. Growth periods may last several months, during which water quality may change irregularly, hence the representativeness of the measurements is rather uncertain. Furthermore, since the sampling locations usually were not routinely monitored sections, it is hard to derive quantitative relationships between pollutant concentrations and their effects. Due to our inability to relate damage to specific pollutant levels, we base the calculation mainly on the area that is being irrigated with wastewater, applying the results from the second survey and some other Chinese studies to estimate the reduction in quantity and quality associated with wastewater irrigation for given crops (Gu 1984; Yang 1984; Chen 2001; Wang 2002; Gao 1997, Sun 2001; Fu 1999).

CAUSAL AGENTS, IMPACT PATHWAY, AND CALCULATION MODEL

Causal Agents

The comprehensive pollution index or weighted comprehensive pollution index (P_c) is usually used to indicate water quality. However, due to limited data, dose-response functions for effects on crops applying these indices cannot be constructed at present.

The second survey of wastewater-irrigated areas distinguished between two kinds of sewage water irrigation: (1) clear water and sewage mixed irrigation (CSMI), and (2) pure sewage irrigation (PSI). Generally speaking, the water used in PSI was of poorer quality and thus more dangerous to crops; water qualities of CSMI, although varying considerably, were less dangerous than PSI to crops. In this project, we calculated damage to farm crops caused by both types of irrigation.

Impact Pathway

Use of polluted water for irrigation affects agricultural production both by reducing the quantity and the quality of output. The reduction in quality is related to two factors: (1) an excess of pollutants in crops, originating from heavy metals or other toxic substances in wastewater, making the crop unsuitable for human consumption; and (2) substandard nutritional quality, with less protein, amino acids, Vitamin C, and other nutrients. For example, rice of poor quality produced more brown and damaged grains. Wheat of poor quality produced less flour and gluten. Vegetables of poor quality have an unpleasant taste and contain more nitrate and nitrite (Gao 1997; Sun 2001; Zhang 1999; Bai 1988). The damage to farm crops caused by water pollution is expressed as a percentage of crop reduction and percentage of production having reduced quality.

Calculation Model

PSI and CSMI areas in each province are obtained from the *Second National Survey Report of Wastewater-irrigated Area* (MoA 2001). As mentioned, economic losses from crop damage caused by water pollution stem from reductions in both quantity (reduced yield) and quality (excess pollutants and substandard nutritional value). Since crops with reduced nutritional quality may or may not have excess pollutants, we calculated the loss due to reduced nutritional quality according to equation. 5.1, i.e. as the mean value of the above two possibilities as presented in equations.5.3 and. 5.4. The effect of introducing equation 5.4 is to avoid a double counting of economic loss otherwise likely to occur when the percentage quantity of a crop that is contaminated (contains pollutant levels above health guidelines) is high. The three equations are described below.

PSI and CSMI areas in each province are obtained from the *Second National Survey Report of Wastewater-irrigated Area.* Losses of farm crops caused by water pollution consist of three parts, which can be calculated with the following formulas. Since crops with excess of pollutants may or may not have reduced nutritional quality, we propose the third loss expressed in equation 5.5 as the mean value of the above two possibilities as presented in equation 5.3 and equation 5.4.

(1) Economic loss due to yield reduction

$$L_1 = \sum_{i=1}^{4} \alpha_{1t} S_i Q_i P_i / 100 \qquad (5.1)$$

(2) Economic loss due to excess pollutants in crops

$$L_2 = \sum_{i=1}^{4} (1 - \alpha_{1i}) \cdot \alpha_{2i} \cdot \beta_{2i} \cdot S_i \cdot Q_i \cdot P_i / 10^4 \qquad (5.2)$$

(3) Economic loss due to nutritional quality decline

$$L_{3u} = \sum_{i=1}^{4}(1-\alpha_{1i})\cdot\alpha_{3i}\cdot\beta_{3i}\cdot S_i \cdot Q_i\cdot P_i/10^4 \quad (5.3)$$

$$L_{3l} = \sum_{i=1}^{4}(1-\alpha_{1i})\cdot(1-\alpha_{2i})\cdot\alpha_{3i}\cdot\beta_{3i} \cdot S_i\cdot Q_i\cdot P_i/10^6 \quad (5.4)$$

$$L_3 = (L_{3u} + L_{3l})/2 \quad (5.5)$$

These three equations are put together below to determine the total economic loss to crops from polluted water irrigation:

(4) Total economic loss from crops irrigated with polluted water

$$L_3 = L_1 + L_2 + L_3 \quad (5.6)$$

where L is the economic cost of reduced agricultural yield and reduced crop quality caused by polluted irrigation water, in tens of thousands of RMB; L_1 is the economic loss due to yield reduction; L_2 is the economic loss due to excess pollutants in crops; L_3 is the economic loss due to nutritional quality reduction; L_{3u} is the economic loss only due to nutritional quality reduction; L_{3l} is the economic loss when the crop both has excess levels of pollutants and decreased nutritional levels; P_i is the market price of crop i, in RMB/kg; S_i is the wastewater irrigated area of crop i (ha), Q_i is the yield per unit area of crop i in clean region, kg/ha; α_{1i} is the fractional quantity reduction of crop i from water pollution; α_{2i} is the fractional quantity of crop i that is contaminated (contains pollutant levels above health guidelines); α_{3i} is the fractional quantity of crop i with nutritional quality decline; β_{2i} is the coefficient of value loss of crop i due to contamination[8]; and β_{3i} is the coefficient of value loss of crop i due to poor quality, determined by the degree of quality decline.

DOSE-RESPONSE COEFFICIENTS AND OTHER PARAMETERS

Areas in China Irrigated with Wastewater

A report that recorded the area irrigated with wastewater from 1949 to 1982 states that the total wastewater-irrigated area was 1398.7 kha in 1982 (Dong 1985). A second report from 1996 to 1999 indicates that the total area was 3639.3 kha (with 1995 as the base year) MoA 2001). During the 13–14 years between the two timeperiods of the studies, the total area increased by 2219.4 kha, that is, by a factor of 1.6. The sewage-irrigated areas in each province from the second survey are listed in Table 5.6, and the yearly increase in the sewage-irrigated area is listed in Table 5.7. (Dong 1985).

Table 5.7 shows that the average annual increase in wastewater irrigation area tended to accelerate during the period from 1949 to 1982. The average increase in the area of wastewater irrigation was only 2.81kha between 1949 and 1963. A sharp increase occurred in the period from 1979 to 1982, reaching 355.13kha. After 1982, the rate of change slowed down.

The increase of wastewater irrigation area is related to three factors:(1) increases in planted area, (2) limited water resources, and (3) available wastewater in China. Given the relative constraints or driving forces of these three factors, we assume the change in area irrigated with wastewater in China is consistent with the Pearl growth function (Pearl and Reed 1920). The general expression for the Pearl growth function is:

$$Y = Y_C/[1 + \mathrm{EXP}(a + bx)] \quad (5.7)$$

where Y is the area irrigated with wastewater; Y_C is the maximum value of the wastewater irrigation area; and x is the year.

When we performed the regression analysis using the Pearl function, we first assumed a value for Y_C, and then performed a regression using

TABLE 5.6 Wastewater-Irrigated Areas by Province (kha)

Region	CSMI	PSI	Total
Beijing	0	13.60	13.6
Tianjin	119.15	114.88	234.0
Hebei	96.68	18.50	115.2
Shanxi (Tai-yuan)	79.34	6.64	86.0
Inner Mongolia	104.67	24.00	128.7
Liaoning	477.1	57.90	535.0
Jilin	0.72	0.00	0.7
Heilongjiang	75.48	13.13	88.6
Shanghai	14.4	0.00	14.4
Jiangsu	71.12	6.30	77.4
Zhejiang	14.73	0.00	14.7
Anhui	638.19	0.27	638.5
Jiangxi	15.05	4.96	20.0
Fujian	0.55	0.05	0.6
Shandong	262.68	85.83	348.5
Henan	670.36	45.48	715.8
Hubei	30.3	11.70	42.0
Hunan	211.28	58.80	270.1
Guangdong	5.91	1.90	7.8
Guangxi	2.25	0.56	2.8
Hainan	0	0	0
Sichuan	55.1	2.85	58.0
Chongqing	3.04	0.07	3.1
Guizhou	5.28	0.00	5.3
Yunnan	10.01	6.30	16.3
Shanxi (Xi-an)	119.74	33.89	153.6
Gansu	28.77	0.36	29.1
Qinghai	6.66	0.00	6.7
Ningxia	4.16	2.00	6.2
Xinjiang	4.97	1.63	6.6
Total	3,127.69	511.60	3639.3

Source: Second National Survey Report of Wastewater-irrigated Area.

available data. In practice, we used data from between 1949 and 1990 as the regression sample points and performed the regression using different Y_C's. Results provided by Eview statistical software are given in Table 5.8. From these values, we obtain the relationship, given by equation 5.2.9, between Y_C and Y_{1995} by regression. Using the observed value for Y_{1995}, the estimated Y_C is obtained.

$$Y_C = -2959 + 1.9497 \quad Y_{1995}$$

$$R^2 = 0.9998 \tag{5.8}$$

According to the second wastewater irrigation area investigation, the Y_{1995} value is 3,639.3kha. Inserted into formula 5.2.8, the appropriate value of Y_C is 4,136. Then the final regression function to predict Y is presented as formula 5.9.

$$Y = 4136/\left[1 + \mathrm{EXP}\left(8.70963 - 0.23253X\right)\right]$$

$$R^2 = 0.96307 \tag{5.9}$$

Using the above equation, the wastewater-irrigated area in 2003 was predicted to be 4,050 kha.

Total Sewage-irrigated Cropland (*S*) and Per Unit Area Yield (*Q*)

Because of lack of more specific data on areas planted with various crops in every wastewater irrigated region, the total area of sewage-irrigated cropland in each province was calculated based

TABLE 5.7 Sewage-Irrigated Area in China, 1949–1995 /kha

Year	1949	1963	1972	1976	1979	1982	1990	1995
Total area	0.7	40.0	93.3	180.0	333.3	1398.7	3333	3639.3
Annual average increase		2.81	5.92	21.68	51.10	355.13	241.79	61.26

Source: Agricultural Environmental Protection Institute, Ministry of Agriculture.

TABLE 5.8 Regression Results with Different Y_c Values

Y_c	A	B	Y_{1995}
3,500	8.869709	–0.25527	3312.67
4,000	8.711862	–0.23506	3563.93
4,500	8.725913	–0.22765	3831.45
5,000	8.771965	–0.22333	4088.74
5,500	8.827719	–0.22041	4332.20

on information from the *China Agriculture Yearbook* (China Agricultural Yearbook 2004). It was assumed that the ratios of areas planted with wheat, rice, corn, and vegetables in wastewater-irrigated areas were the same as for the planted areas as a whole within each province, province-municipality, or autonomous region. So for each province we get:

$$S_i = c_i \cdot S_t \qquad (5.10)$$

where S_i is the area planted with crop i in a wastewater-irrigated region, in kha; c_i is the fraction of area planted with crop i to the total planted area in one province, province-municipality, or autonomous region; and S_t is total wastewater-irrigated area in the province, kha.

The yield per unit planted area (Q) is the average yield of different provinces or municipalities, calculated from information in the *China Agriculture Yearbook 2004.*

Identification of α_{1i}, α_{2i}, and α_{3i}

The term α_{1i} represents the percent by which the quantity of crop i has been reduced as a result of environmental pollution.

Our determination of crop loss caused by wastewater irrigation was mainly based on data from field experiments, and the final estimates are conservatively adjusted results of these data. The documents of the first *National Agricultural Environmental Quality Investigation in Wastewater Irrigation Areas* (MoA 1984) showed that CSCI would not lead to a reduction in crop yield, but, on the contrary, would increase the yields to some extent, about 450–750 kg per hectare. This is mainly due to the presence of nutrient elements such as N, P, K, Cu, and Zn, which are essential to crop growth, in wastewater. On average, the content of N in wastewater is 15.2 mg/L; P_2O_5 is 2.8mg/L; and K_2O is 2.4 mg/L.

Other studies, however, indicate that PSI will lead to yield reduction. According to field experiments that were made in wheat lands irrigated with wastewater from the Liangshui River, the Tonghui River and the Wanquan River by Bai Ying et al. (1988), there were 11 cases of yield reduction among 15 cases, and the reduction percentage was about 8.0–17.1 percent. Similar experiments were conducted in the Gaobeidian region of the Tonghui River and the Yizhuang region of the Lianghe River, where there were 20 years of sewage-irrigation history. Yields of both wheat and rice grown in relatively unpolluted soils in the sewage-irrigated area decreased 10 percent compared to clean water-irrigated areas; yields of wheat and rice grown in polluted soil irrigated with sewage decreased even more, 40.6 percent and 39.0 percent respectively, compared to clean irrigation areas.

Chang et al. (2001) also indicates that sewage-irrigation can cause production reduction. Their study proposes expressing reductions as a function of the comprehensive water pollution index (P), when P >1.0 sewage irrigation caused yield reductions of 10 percent for wheat and 30 percent for rice and vegetables.

Sewage irrigation can affect the growth of roots and seedlings in rice crops and tillers in wheat crops. The height, leaf area, and dry matter can be reduced. Because of reduced leaf surface area, the photosynthesis of wheat is reduced. All these factors directly affect crop production. In conclusion, the negative effects on the yield of wheat and rice mainly occur as a reduction in the number of ears per unit area, number of seeds per ear, and seed weight. The clean water in CSCI can alleviate the damage, however, and generally does not lead to large yield reductions (Bai 1988).

Based on the above considerations, we suggest using the a_{1i} values given in Table 5.2.5 until more information becomes available.

(2) Identification of α_{2i}—percentage quantity reduction of crop *i* from exceedance of pollutant criteria

As mentioned, the term α_{2i} represents the percent of crop *i* that contains levels of pollutants above health guidelines, which is probably the most severe effect of wastewater irrigation. The accumulation of harmful pollutants in farm crop products renders large amounts of products unsuitable for human consumption or even useless. The results of the second survey of wastewater irrigation show that the main pollutants in wastewater-irrigated areas were heavy metals, such as Hg, Cd, Pb, Cu, Cr, and As. The primary pollutants that exceed allowable thresholds in wheat are Hg, Cd, Pb, and Cu; in rice, Hg, Cd, and Pb; in corn, Cd and Pb; and in vegetables, Hg, Cd, and As. The extent to which pollutants in the four crops exceed allowable thresholds is shown in Table 5.9. The field experiments showed that contents of NO_3^- and NO_2^- in vegetables from sewage-irrigated areas were considerably higher than those prevailing in clean water-irrigated regions (Bai 1984).

The data in Table 5.9 show that the pollution levels in 17 percent and 29 percent of the wheat crops in the CSPI regions and PSI regions respectively exceeded allowable thresholds; for rice, the values were 42 percent and 51 percent; for corn, 16 percent and 18 percent; and for vegetables, 12 percent and 27 percent. It is clear from the table that the damage to crops is serious for both CSCI and PSI. For all crops, those irrigated with PSI exceeded allowable levels of pollution by a far greater rate than those irrigated with CSCI. In this project, we estimated only economic losses caused by yield reduction, excess pollutants, and poor quality, and did not include losses due to ecological environmental degradation and damage to human health.

TABLE 5.9 Ratio of Crops Exceeding Standards from 2nd Survey of Sewage Irrigation

Crops	Type of Sewage Irrigation	Total Reported Yield /t	Yield Failing to Meet Pollution Standards /t	Percentage of Yield Failing to Meet Pollution Standards
Wheat	CSMI	305,466	52,218	17
	PSI	291,540	84,603	29
Corn	CSMI	789,766	128,608	16
	PSI	242,912	43,608	18
Rice	CSMI	187,480	79,442	42
	PSI	182,267	92,321	51
Vegetable	CSMI	188,723	22,838	12
	PSI	160,580	42,755	27

Source: Authors calculation.

The term a_{3i} refers to the reduction in useful yield of crop *i* from quality decline due to reduced nutrient content

The results of both field experiments and surveys show that wastewater irrigation results in more brown and damaged rice grains, some even with disagreeable tastes. Wastewater irrigation causes low gluten in wheat and low flour production. The results of field experiments showed that the contents of protein and amino acids in wheat produced from wastewater-irrigated areas were lower than in clean-water irrigated areas.

Suspended substances in wastewater apparently affect soil porosity, lowering activity and respiration of wheat roots and leading to a lower protein content. Rice belongs to the helophytes; the roots get oxygen not only from air in soil but also from the atmosphere through leaves and stems, so protein content is not affected.

The effects of wastewater irrigation on production and quality of farm crops are summarized in Table 5.10.

Identification of β_{2i} and β_{3i}

The term β_{2i} represents the price-loss coefficient for crops that exceed allowable pollution

TABLE 5.10 Effects of Wastewater Irrigation on Production and Quality of Farm Crops (%)

Crops	Type of Sewage Irrigation	Percent by Which a Crop is Reduced by Environmental Pollution (a_{1i})	Percent by Which a Crop is Reduced by Excess Pollutants (a_{2i})	Percent by Which a Crop is Reduced by Decline in Nutrient Content (a_{3i})	Major Pollutants
Wheat	CSMI	0	17	0	Cd, Hg, Pb,
	PSI	10	29	10	Cu, As, Zn, F
Corn	CSMI	0	16	0	Cd, Hg, Cu,
	PSI	10	18	10	Pb, As
Rice	CSMI	0	42	0	Cd, Hg, Pb, Cu,
	PSI	20	51	5	Cr, As, F
Vegetable	CSMI	0	12	5	Cd, Hg, Pb, Cr,
	PSI	15	27	15	Cu, F, NO^{3-}

Source: Authors calculation

thresholds. If the thresholds are exceeded, the crops become inedible. Wheat, rice, and corn that fail to meet quality standards can be put into industrial use, with a value half that of products that do meet the standards, which means β_{2i} will be 0.5. However, the vegetables become waste and β_{2i} will be 1.

The terms β_{3i} represents the price-loss coefficient for crops with reduced quality due to nutrient reduction. Market prices indicate that the price of crops of this kind is moderately lower than that of high quality products. We set β_{3i} of wheat, corn, and rice to 0.8, and that of vegetables to 0.7.

Prices of agricultural products

According to the data on the national agricultural information website issued at the beginning of 2004, the average prices (wholesale price) of the above four kinds of crops in 2003 were as follows: rice 1.20RMB/kg, corn 1.15RMB/kg, wheat 1.14RMB/kg and vegetables 1.42RMB/kg.

RESULTS

Effects of wastewater irrigation on agriculture include production reduction, poor quality crops, and crops that fail to meet standards of allowable pollution levels, as well as deterioration of agro-ecological environments. The effects on agro-ecological environments include soil pollution of farmland, and destruction of soil structure and groups of soil microorganisms. If the agro-ecological environment is degraded, recovery is very difficult to achieve. This project does not include agro-ecological environmental deterioration; it only calculates economic losses of wheat, corn, rice, and vegetables caused by wastewater irrigation. The calculation results are shown in table 5.11. Table 5.12 presents the economic mid-loss of every province caused by wastewater irrigation in 2003.

Tables 5.11 and 5.12 show that the direct economic losses for crops in 2003 was about 6.7 billion RMB. The losses of four crops are: wheat, 0.4 billion RMB; corn, 0.5 billion RMB; and rice, 1 billion RMB. Loss of vegetables dominates with about 73.5 percent of the total. Economic loss caused by failure to meet pollution standards is 5.2 billion RMB, about 78.5 percent of the total agricultural economic loss. By adding the loss caused by quality decline, 85.1 percent of the total loss is obtained. The high and low total crop losses are respectively

TABLE 5.11 Economic Losses of Wheat, Corn, Rice, and Vegetables in 2003 (10,000 RMB)

Loss		Wheat	Corn	Rice	Vegetables	Total
Output reduction		4,463	7,188	8,211	79,630	99,491
Excessive pollution levels		31,833	37,247	86,190	368,696	523,967
Poor nutrient quality		687	1,177	245	42,220	44,329
Total	37,099	45,729	94,729	494,861	672,419	672,419
	36,983	45,613	94,646	490,546	667,787	667,787
	36,867	45,496	94,562	486,231	663,155	663,155

TABLE 5.12 Economic Mid-Loss Caused by Wastewater Irrigation by Province in 2003

	Economic Loss /10^4 Yuan					Agricultural	Percentage to
Regions	Wheat	Corn	Rice	Vegetable	Total	Output/ 10^8 Yuan	Agricultural Output
Beijing	252	442	37	14,925	15,657	88.8	1.76
Tianjin	3,496	4,982	1,339	121,851	131,669	88.2	14.93
Hebei	1,846	1,686	186	21,222	24,940	958.3	0.26
Shanxi (Tai-yuan)	629	1182	13	6,203	8,028	249.5	0.32
Inner Mongolia	271	2,441	356	4,551	7,620	336	0.23
Liaoning	109	14,678	17,816	76,397	109,000	497.3	2.19
, Ji-in	0	25	13	25	63	438.3	0.00
Heilongjiang	98	1,084	2,476	3,038	6,697	502.9	0.13
Shanghai	27	11	755	3,196	3,989	98.2	0.41
Jiangsu	764	313	4,673	10,129	15,879	981.2	0.16
Zhejiang	11	12	854	1,942	2,819	529.4	0.05
Anhui	5,051	2,464	22,642	25,791	55,947	617.9	0.91
Jiangxi	0	0	27	60	88	466.8	0.00
Fujian	2	3	2,088	1,353	3,446	383.7	0.09
Shandong	7,159	5,585	1,050	87,234	101,027	1,599.3	0.63
Henan	14,059	7,168	5,937	64,456	91,621	1,137.7	0.81
Hubei	126	139	3,133	6,064	9,462	733.4	0.13
Hunan	84	597	25,530	24,981	51,193	671.7	0.76
Guangdong	0	11	692	1,310	2,013	851.7	0.02
Guangxi	0	10	186	247	443	500.8	0.01
Hainan	0	0	0	0	0	152.7	0.00
Sichuan	8	18	119	176	321	270.1	0.01
Chongqing	302	342	2,521	3,721	6,887	804.7	0.09
Guizhou	10	39	98	169	316	275.5	0.01
Yunnan	67	198	652	934	1,852	433.9	0.04
Xizang						25.3	0.00
Shanxi (Xi-an)	2,133	1,823	1,299	8,120	13,375	334.4	0.40
Gansu	247	179	10	1,479	1,915	275.8	0.07
Qinghai	52	0	0	238	291	29.7	0.10
Ningxia	79	98	101	332	610	54.1	0.11
Xinjiang	100	81	41	400	622	482.8	0.01
Total	36,983	45,613	94,646	490,546	667,787	14,870.1	0.45

6.7 and 6.6 billion RMB; the gap between them is about 1.39 percent to the medium estimation.

The above data fully reflect that large amounts of polluted crops have been produced from wastewater irrigation areas every year. These inferior products constitute a great threat to food safety and human health if they enter markets and the potential negative effect on human health may greatly exceed the direct economic loss. Thus, the phenomenon merits high attention by the related sectors.

China is a country lacking freshwater resources and water for agricultural irrigation is in extreme shortage. Due to the successive droughts for many years in northern China, north of the Yangtze River, wastewater has become an important resource for agricultural irrigation and the area using wastewater irrigation has continued to expand in recent years. In China, about 480 hundred million tons of wastewater are discharged every year, as discussed elsewhere in this report (SEPA 2004). For better utilization of water and fertilizers in wastewater, it is important to seek better treatment and strengthened control of wastewater to meet the standards of water quality for farm irrigation. One should also strive to gradually decrease wastewater-irrigated areas.

UNCERTAINTIES

The uncertainties in estimating wastewater irrigation effects and the economic cost of agricultural production reduction due to pollution are mainly related to the following issues:

(1) *Selection of dose-response functions.* First, the appropriate dose-response function must be based on a scientific methodology and a strict research process. In current research literature concerning the relationship between water pollution and the quantity and quality of agricultural products, methods are not identical, which causes some difficulties in the selection of dose-response functions.

Second, dose-response functions for wastewater irrigation are derived for a particular case, for example, a particular region, and a given mix of pollutants, therefore the results are not necessarily generalizable.

Water pollution may damage agricultural crops in several ways and systematic research is lacking. The effect estimates in this guideline are mainly based on data and analysis of the second national wastewater irrigation investigation. As there is no data on extent of damage of agricultural crops and corresponding irrigation water quality, dose-response functions have not been established between the composite water quality index and quantity and quality of agricultural crops. Instead effects are estimated simply for two types of irrigation: pure wastewater irrigation (PSI) and mixed clear and waste water irrigation (CSMI). Clearly, the uncertainties in the estimates are large.

(2) Determination of value loss coefficients
The β-values given above were estimated from market investigations. However, there are clearly large variations in degree of damage and demand for the products of reduced quality so the β values are highly uncertain. Uncertainty in the β values is among the main sources of uncertainty in the final result.

5.3

Fishery Loss

Due to the episodic nature and easily measurable effects of acute pollution, its impact on fisheries is much better understood than the effects of chronic pollution. Analyses conducted by the Ministry of Agriculture and SEPA estimated that fishery losses due to acute pollution accidents in 2003 amounted to over 4.3 billion RMB, including 713 million RMB in direct economic losses and more than 3.6 million RMB in indirect losses (MoA and SEPA 2003). While not insignificant, these figures may greatly underestimate the total economic cost of fishery loss due to pollution. First, chronic pollution costs are likely to be much higher than the acute. Secondly, the methodology employed in these studies—such as the application of a rule that stipulates that the indirect costs cannot be higher than three times the direct—may further underestimate the true cost of pollution.

Water pollution can damage both marine fisheries and inland waters fisheries. The marine fishery zone includes coastal sea areas and offshore: the inland waters fishery includes cultivation in rivers, lakes, estuaries, and reservoirs. These fisheries can suffer both acute and chronic toxic effects of pollution. In marine waters, acute toxicity mainly refers to the extensive death of aquatic animals and plants caused by red tide (see box 5.3). In inland waters, acute toxicity is generally triggered by excessive discharge of high concentration pollutants in inland waters. Chronic toxicity of fisheries in both cases results from the long-term accumulation of pollutants and mutagenic substances in the water bodies. This pollution-related damage to the fisheries results in a loss of production, which can be described in terms of direct and indirect economic loss. Direct economic loss means that pollution sources contaminate the fishing zone, killing or damaging valuable and/or rare and endangered aquatic wildlife such as fish, shrimp, crab, shellfish, and algae. Indirect economic loss refers to the possible loss of fishery production caused by reduced reutilization of natural fishery resources, reduced capacity to reproduce, and decreased breeding grounds. The direct and indirect losses have been estimated by the fishery supervision and management agency (MoA 1996).

Acute pollution accidents may cause a high death rate, which tends to attract public attention. When large numbers of aquatic animals and plants die over a short time, the ensuing economic loss can be more easily measured and calculated. However, the fish morbidity rate resulting from chronic damage caused by prolonged exposure to polluted water may be more serious, with the extent of damage depending on the degree of pollution. Because chronic damage takes place over a long period of time and is not easily observed, it tends to be ignored. Furthermore, the lack of systematic research on the exposure–response relationship for aquatic animals and plants in the polluted water body makes it difficult to evaluate the loss due to chronic damage. With respect to both the quality and quantity of damage, chronic water pollution may result in greater losses than those caused by acute toxicity.

BOX 5.3 Background Knowledge

Red Tide and Its Damage Mechanisms

Red tide occurs when environmental conditions in the ocean change and cause a bloom of phytoplankton algae, which in turn changes the color of the water. Not all red tide is red. The actual color of waters experiencing red tide depends on the species of algae that blooms. Red tide can have either natural or anthropogenic origins. Natural causes include shifts in climatic factors, sea temperature, salinity, and seawater exchange. Human-induced red tide generally stems from marine aquaculture and pollution. Red tide causes severe damage to the marine ecosystem and even endangers human health through five damage mechanisms:

(1) *Mucus excretion.* The algae excrete mucus, which adheres to the gills of marine animals, impeding respiration and causing them to die by suffocation.
(2) *Chemical excretion.* The algae excrete chemical substances (such as ammonia, hydrogen sulfide) that harm the water body and poison other organisms.
(3) *Toxin production.* The algae produce toxins that directly poison cultivated animals and plants and/or are transferred through the food chain to damage human health by poisoning.
(4) *Oxygen depletion.* The algae use the oxygen in the water body or cause water to contain a great deal of hydrogen sulfide and methane so that cultivated organisms may die from oxygen depletion or poisoning.
(5) *Absorption of solar rays.* The algae absorb sunshine and shade the sea surface, causing aquatic plants to die from insufficient sunshine, which could further reduce fish populations.

Chronic pollution not only directly causes excessive levels of pollutants in aquatic animals and plants, but may also cause changes in the biological community affecting the ecological balance of the whole water body. Furthermore, it may bring about so-called secondary pollution loss problems when humans eat the aquatic products with pollutants exceeding standards.

To quantitatively evaluate pollution-induced fishery loss, the Chinese Ministry of Agriculture issued the *Regulations on Calculation Method of Fishery Loss Caused by Pollution Accidents in Water Area* in 1996. The calculation method takes into account the type of water (marine or inland), hydrographic conditions, the size of the polluted area, and the type of damaged resources. The regulations provide a basis for scientific and rational calculation of fishery loss caused by pollution accidents, as well as guidelines for how to handle such accidents. The *China Fishery Ecological Environmental Condition Bulletin,* issued jointly by the Ministry of Agriculture and the State Environmental Protection Administration each year, publishes the fishery loss caused by pollution accidents as calculated according to the regulations. But as described above, due to the lack of research on the exposure–response relationship between pollution severity of the water body and fish growth and reproduction, chronic damage still cannot be estimated. Therefore, fishery loss in the following only refers to fishery damage caused by acute pollution accidents.

In addition, it should be noted that overfishing and poor fisheries management practices, as well as irrational water resource development and utilization, can also affect the sustainability of fish resources and change ecological environments and migration routes for fishes. In some cases, aquatic animals and plants may be pushed toward extinction. Such losses are caused by poor management and are not evaluated in this project.

ESTIMATED FISHERY LOSS DUE TO ACUTE POLLUTION EPISODES

Estimation Method

Direct economic loss includes loss of aquatic products, loss of additional pollution protection facilities, loss of fishing gear, pollution removal cost, and actual cost of evidence collection and identification by monitoring departments. The aquatic product loss is quantified according to the market retail price provided by the local business administration department. Loss of aquatic products includes the quantity of fish killed by exposure to toxic pollutants, the quantity that has apparent toxic symptoms but still can survive, and the quantity that has been rendered inedible due to pollution. The loss includes both finished products and semi-finished products, as well as loss of offspring. The quantified loss is expressed in terms of loss of finished products, with conversion factors for offspring and semi-finished products determined by the Fishery Supervision and Management Agency according to different species and actual local situation. For fish cultivation in cages or in rice fields, loss is quantified as follows:

$$\textit{Loss of aquatic product} = \textit{local market price} \times \textit{loss quantity}$$

The cost of death of a parent strain and original seed of cultivation species shall be set 50 to 500 percent higher than the general commodity price, depending on its degree of importance. The specific price is determined by the Fishery Supervision and Management Agency.

The costs of pollution protection facilities, fishing equipment, pollution removal, and evidence collection and identification by the Fishery Supervision and Management Agency are calculated according to actual expenditures.

Economic loss for natural fishery resources is determined in accordance with the local resource situation by the Fishery Supervision and Management Agency, but shall not be less than three times the direct economic loss obtained from the estimated reduction in amount of aquatic products.

Estimation Result

According to the *China Fishery Ecological Environmental Condition Bulletin,* the overall ecological situation for fisheries in China was good in 2003, but at the local level, some waters were still seriously polluted by nitrogen, phosphorus, petroleum, and some heavy metals. In 2003, fishery pollution accidents occurred, resulting in fishery losses totaling over 4.3 billion RMB—including 713 million RMB in direct economic losses and more than 3.6 billion in indirect losses related to measurable damage to natural fishery resources, of which 896 million RMB was from inland waters and 2.7 billion RMB from natural ocean fishery resources.

Due to the difficulties in estimating the fishery loss from the chronic effects from pollution, the stated loss by the *Bulletin* may be only a small fraction of the total loss, since chronic effects are believed to be more serious than acute effects. Another big estimation bias is from the method to evaluate the indirect loss. The regulation that the economic loss for natural fishery resources shall not be less than three times the direct economic loss is a rather subjective judgment.

Endnotes

1. Where no reference is given, data in this chapter is from personal communications with staff at the Ministry of Water Resources, China.
2. The six are Beijing, Tianjin, Hebei, Shanxi, Shanghai and Ningxia.
3. Lioaning, Jiangsu, Shandong, Henan and Gansu.
4. Data from 2005 indicate that even Shandong is below 500, while Gansu is above 1000. (NBS, 2006). 2005 was also an average year in terms of nationwide water resources.
5. Worse than class IV in the groundwater standard (GB/T14848-93).
6. Worse than class III in the same standard.
7. The range 2.1–5.2 produced by He and Chen—depending on river basin—may also be compared with values used by the World Bank (2001)—depending on final consumption. The World Bank (2001) values range from 0.8 in agriculture to 6 in urban industry.
8. Because not all substandard crops are discarded but used for other purposes with lower quality criteria such as fodder and industrial raw materials, this factor is introduced.

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6

NON-HEALTH IMPACTS OF AIR POLLUTION

6.1

The Impact of Acid Rain and SO_2 on Crop Loss

Dose-response functions estimated from studies in the 1980s and 1990s for different crops and regions within China show that crop losses due to SO_2 and acid rain accounted for about 30 billion RMB in 2003. About 80 percent of this economic loss was associated with lower vegetable yields, and nearly half of the total cost was incurred by only three provinces—Hebei, Hunan, and Shandong. This study did not estimate the cost of acid rain on forests due to the lack of reliable exposure-response functions and in order to avoid attributing a cost, which if based on timber loss alone would significantly underestimate the true cost.

Vegetation damages may be caused by direct exposure to gaseous or particulate air pollutants or indirectly through soil acidification. Direct damage from SO_2 emissions is very likely in some regions. Other possible causes of damage include high concentrations of ozone and other photo-oxidants and, in some areas, hydrogen fluoride in the air. In many countries, ozone is the main damaging agent; this is an increasing factor also in China (Aunan et al. 2000; Wang and Mauzerall 2004). However, due to scarcity of monitoring stations, especially outside cities, ozone damage is not considered in this report. Indirect effects due to soil acidification may be from elevated levels of toxic aluminum in soil water, increased leaching of plant nutrients (particularly magnesium) from soils, or reduced availability of phosphorus. Acidic mist or acidic cloud water can reduce tolerance of certain species to cold. In most pristine forests, increased deposition of nitrogen will increase growth rates, but if nitrate deposition becomes too high it may result in damage due to soil acidification, lack of other nutrients, or increased sensitivity to other stress factors. In many cases, vegetation damage is likely a combined effect of anthropogenic and natural stressors (e.g., drought, frost, and pests).

In this report, only effects of SO_2 and acid rain are considered. There are large uncertainties in estimates of vegetation effects. Here we will propose a procedure for estimating effects on crops mainly based on Chinese studies. We warn against quantification of effects on forests at the present state of knowledge, but in Appendix X (to be included) we give tentative equations based on Chinese studies.

CAUSAL AGENTS AND IMPACT PATHWAY

There are many studies on effects of SO_2 and acid rain on crops. However, the results are sometimes in conflict, and establishing dose-response functions—especially for acid rain—is difficult. A recent study showing clear effects of acid rain has been carried out in India (Singh and Agrawal 2004). However, in this report we will use results of Chinese studies.

Acid rain and SO_2 damage to crops can be divided into acute injury and chronic injury. Acute injury implies that the leaves get clear signs of injury within a short time due to contact with acid rain or SO_2. This kind of injury generally appears when the pollution levels are very high. Long-term exposure to lower pollution levels may cause chronic injury, for example due to changes in chlorophyll or pigment. This destroys the normal activity of the cells, causing cell death and/or symptoms such as early loss of leaves.

In addition, though pollutants may affect soil conditions, e.g. cause soil acidification, it is not clear how much such indirect effects may reduce yields. So, the possible indirect loss through soil changes caused by acid rain is not included in this ECM-edition.

Dose-Response Relationships

There were some studies in China on dose-response relationships between SO_2/acid rain pollution and crop yields in the 1980s and 1990s. In one important experiment, crops cultivated in pots were exposed to various levels of SO_2 and acid rain pollution. The acid rain used in the experiment simulated as far as possible natural acid rain in South China in concentrations of SO_4^{2-}, NO_3^-, Ca^{2+}, Mg^{2+}, K^+, Na^+, and Cl^-. The weight ratio SO_4^{2-}/NO_3^- was 9:1. The ratio is likely to be lower now (**see Annex A for discussion**). The watering regime mimicked the conditions for the growth periods in 1955–85 for Nanning City in Guangxi Province—that is, using average rainfall each month and average number of days with rainfall greater than 10mm (moderate rain).

The experimental outputs of crops cultivated in pots exposed to SO_2 and acid rain are shown in Table 6.1.

Based on the above data, we obtained the dose-response relationships between SO_2/acid rain pollution and crop yields (see table 6.2).

Limits in SO_2 concentration and pH identify the type of pollution in the area. When $[SO_2] \geq 0.04 mg/m^3$ and pH ≤ 5.0, the crops are under combined acid rain/SO_2 pollution stress. For loss estimation, we suggest the set of functions in the right column of Table 6.2. When $[SO_2] \geq 0.04 mg/m^3$ and pH > 5.0, only SO_2 has an effect on crops, and the functions in the left column of Table 6.2 can be used. When $[SO_2] < 0.04 mg/m^3$ and pH ≤ 5.0, the crop lossesare only from acid rain and can be estimated with the functions in the middle column of Table 6.2.

Valuation Model

$$C_{ac} = \sum_{i=1}^{n} a_i P_i S_i Q_{0i} / 100 \qquad (6.1)$$

where: C_{ac}—Economic cost of crop yield reduction caused by air pollution, 10,000 RMB;

TABLE 6.1 Yields of Crops Cultivated in Pots Exposed to SO_2 and Acid Rain

Crops	SO_2 Only	Acid Rain Only	SO_2 and Acid Rain
Rice	$Y = 26.01 - 2.85X_1$	—	$Y = 26.61 - 4.82 X_1 + 0.049 X_2$
Wheat	$Y = 23.52 - 6.33 X_1$	$Y = 17.20 + 1.17 X_2$	$Y = 17.84 - 7.14 X_1 + 1.04 X_2$
Barley	$Y = 34.11 - 12.22 X_1$	$Y = 27.29 + 1.55 X_2$	$Y = 25.84–15.51 X_1 + 1.53 X_2$
Cotton	$Y = 30.60 - 7.70 X_1$	$Y = 24.07 + 1.26 X_2$	$Y = 22.15–8.84 X_1 + 1.62 X_2$
Soybeans	$Y = 40.82 - 11.75 X_1$	$Y = 34.68 + 1.12 X_2$	$Y = 30.57–13.24 X_1 + 1.95 X_2$
Rape	$Y = 31.12 - 15.81 X_1$	$Y = 16.85 + 2.71 X_2$	$Y = 19.4–13.02 X_1 + 1.83 X_2$
Carrots	$Y = 105.58 - 56.97 X_1$	$Y = 54.96 + 9.67 X_2$	$Y = 71.03–41.82 X_1 + 5.22 X_2$
Tomatoes	$Y = 92.70 - 34.67 X_1$	$Y = 72.82 + 3.78 X_2$	$Y = 72.95–31.96 X_1 + 2.60 X_2$
Kidney beans	$Y = 43.69 - 30.14 X_1$	$Y = 9.00 + 6.39 X_2$	$Y = 22.90–30.11 X_1 + 3.01 X_2$

Source: Authors Calculations.
Note: Y—Crop yield; X_1—SO_2 concentration in mg/m^3; X_2—pH value.

TABLE 6.2 Dose-Response Relationship Between SO_2/Acid Rain Pollution and Crop Yield Applied in ECM

	Percentage Yield Reduction (%)		
Crops	Pollution by SO_2 (mg/m^3)	Pollution by Acid Rain (pH value)	Combined Pollution of SO_2 and Acid Rain (mg/m^3), pH Value
Rice	$0.1096 X_1$		$0.0292 + 0.1793 X_1 - 0.00182 X_2$
Wheat	$0.2691 X_1$	$0.2759 - 0.0493 X_2$	$0.2461 + 0.3017 X_1 - 0.043949 X_2$
Barley	$0.3583 X_1$	$0.2413 - 0.0431 X_2$	$0.249 + 0.4508 X_1 - 0.044466 X_2$
Cotton	$0.2516 X_1$	$0.2267 - 0.0405 X_2$	$0.2906 + 0.2831 X_1 - 0.051886 X_2$
Soybeans	$0.2878 X_1$	$0.1532 - 0.0273 X_2$	$0.2632 + 0.3191 X_1 - 0.047 X_2$
Cole	$0.508 X_1$	$0.4739 - 0.0846 X_2$	$0.3457 + 0.4392 X_1 - 0.061724 X_2$
Carrots	$0.5396 X_1$	$0.4963 - 0.0886 X_2$	$0.2916 + 0.4171 X_1 - 0.052064 X_2$
Tomatoes	$0.374 X_1$	$0.2252 - 0.0402 X_2$	$0.1664 + 0.3652 X_1 - 0.029711 X_2$
Kidney beans	$0.6899 X_1$	$0.799 - 0.1427 X_2$	$0.424 + 0.7574 X_1 - 0.075712 X_2$
Vegetables	$0.5345 X_1$	$0.481 - 0.0905 X_2$	$0.294 + 0.5132 X_1 - 0.0525 X_2$

Source: Authors Calculations.
Note: X_1—concentration of SO_2, X_2—pH value; The coefficients in the dose-response relationships for vegetables are average values for carrots, tomatoes, and kidney beans.

P_i—Price of crop i, RMB/kg;
S_i—Planted area of crop i, 10^4 ha;
Q_{0i}—Output per unit area of crop i in clean region, kg/ha;
a_i—Reduction rate of crop i due to pollution (exposure-response relation), %;
n—Number of crop types, $n = 6$.

Parameter Sources

TABLE 6.3 Parameters Used in Valuation Model for Crop Reduction by Acid Rain/SO_2 Pollution and Their Sources

Parameters	Unit	Data Sources	Geographical Resolution
S_i: Planted area of crop i	10^4Mu	Agricultural Statistics Yearbook	City
Q_i: Production of unit area of crop i in clean region	kg/Mu	Agricultural Statistics Yearbook	Province
P_i: Price of crop i	RMB/kg	Agricultural StatisticsYearbook	Nation
α_i: Reduction rate of crop i due to pollution, $a = f(X_1, X_2)$	%	Dose-response functions, Table 6.2	City
X_1: Concentration of SO_2 in planted areas	mg/m^3	Environmental monitoring data	City
X_2: PH of rain in planted areas		Environmental monitoring data	City

Source: Authors Calculations.
Note: 1Mu = 1/15 ha

Valuation Result

According to the statistics yearbooks in 2004 of all provinces, there are only 16 provinces, autonomous regions, and provincial cities for which data of different agricultural products are given so that losses can be calculated at the city level. In other areas, data for different agricultural products are only available for the whole province. This implies that we can only calculate pollution losses in these areas on the province level. Annual average SO_2 concentrations and annual pH values were calculated from values for cities. Using province-wide average concentrations of SO_2 and pH may disguise crop damage in pollution-intensive parts of a province and lead to underestimation of crop damage. We correct for a possible underestimation by increasing the pH limit in province-wide data from 5.0 to 5.6. It is generally agreed that above 5.6 no yield reduction occurs, but between 5.0 and 5.6 the functions of Table 6.2 usually show some damage. If the suggested method yields negative economic losses, these are set to zero. The results are given in Table 6.4.

The results show that the economic losses in agriculture caused by SO_2 and acid rain pollution of China in 2003 were about 30 billion Yuan. About 80 percent of the total loss is due to impacts on vegetables. By region, 21 percent of the total loss is from crop loss in Hebei, 12 percent in Hunan, and 11 percent in Shandong.

UNCERTAINTIES

Although there are dose-response relationships for carrots, tomatoes, and kidney beans, only the aggregate total output of vegetables is given in the agricultural statistical yearbooks. In the model, we therefore derive a dose-response function for vegetables from the arithmetic means of the coefficients in the dose-response functions for carrots, tomatoes, and kidney beans. If yield data can be obtained, economic losses of carrots, tomatoes, and kidney beans can be calculated separately.

In the evaluation model, crop loss is calculated as a percentage reduction in yield per unit area attributable to the pollution. We apply the present yield per unit area to calculate a "hypothetical yield" under clean conditions in the given area, thus enabling an estimation of the crop loss. However, several factors influence the yield, such as land fertility and climate. These factors may enhance or reduce the effects of pollution, and the estimate of the hypothetical yield under clean conditions thus becomes very uncertain.

Other main uncertainties in the results obtained by the described model originate from the following aspects:

1) The dose-response relationships in the proposed model have been obtained from pot experiments, not from field studies. Results from other studies show quite large variations and different results for different cultivars. The pot experiments simulated climatic conditions of South China, and there are especially large uncertainties implied when the relationships are applied to estimate crop loss in the northern provinces.
2) While most crops are grown in rural areas, monitoring data are generally only available for urban areas. Using data for SO_2 concentrations and pH from urban areas will likely lead to overestimated crop losses. The error is likely to be most severe for SO_2, which varies more than pH with the distance from the cities. Limited monitoring data are available for rural areas in China. However, there are reasons to believe that the level of SO_2 is substantially enhanced in large rural areas due to extensive use of coal in town and village enterprises and as a main household fuel for

TABLE 6.4 Crop Losses Caused by SO_2 and Acid Rain Pollution of China, 2003 (10,000 Yuan)

	Economic Losses of Crops						
Regions	Rice	Wheat	Rape Seed	Cotton	Soybeans	Vegetables	Total Economic Losses
Beijing	8	341	0	78	162	24,060	24,649
Tianjin	55	807	0	2,633	391	34,413	38,299
Hebei	777	47,120	0	26,428	7,573	554,168	636,067
Shanxi/t	28	13,179	270	5,841	6,999	141,786	168,102
Neimeng	266	1,078	1,367	89	3,576	23,837	30,214
Liaoning	1,061	84	4,737	42	2,708	58,876	67,508
Jilin	233	0	0	0	1,485	8,302	10,021
Heilongjiang	1,173	9	0	0	3,943	10,016	15,142
Shanghai	465	97	306	18	80	15,030	15,997
Jiangsu	0	1,700	1,621	904	503	0	4,727
Zhejiang	0	1,226	8,218	1,346	3,594	178,582	192,966
Anhui	0	56	1,749	1,467	126	4,712	8,110
Fujian	0	34	73	0	476	0	583
Jiangxi	4,715	117	5,506	1,029	1,424	66,710	79,501
Shandong	385	17,024	117	11,853	2,201	309,677	341,257
Henan	335	29,423	1,919	5,282	2,601	155,366	194,926
Hubei	10,784	6,045	11,969	6,622	2,195	99,042	136,658
Hunan	79,245	982	16,609	13,955	9,230	241,061	361,082
Guangdong	0	24	55	0	490		568
Guangxi	38,553	33	362	18	2,031	69,310	110,307
Hainan	0	0	0	0	0	0	0
Chongqing	24,316	5,832	5,914	0	6,155	111,780	153,997
Sichuan	29,068	15,945	19,004	413	9,454	160,822	234,706
Guizhou	4,530	1,697	5,769	16	2,495	40,351	54,858
Yunnan	0	0	0	0	0	0	0
Xizang	0	0	0	0	0	0	0
Shanxi/x	155	14,732	1,719	4,483	1,312	41,149	63,550
Gansu	24	4,267	1,403	1,678	1,704	33,621	42,697
Qinghai	0	0	0	0	0	0	0
Ningxia	307	1,446	13	0	458	8,271	10,496
Xinjiang	38	649	13	516	61	2,852	4,129
TOTAL	**196,521**	**163,945**	**88,714**	**84,713**	**73,429**	**2,393,794**	**3,001,117**

Source: Authors Calculations.

many people. In northern China, the fact that we use annual values instead of averages for the growing season is likely to lead to overestimated crop losses because the pollution level usually is higher during winter.

3) When lacking yield data on municipal levels, the use of provincial averages combined with environmental monitoring data on city levels introduces uncertainties.

FOREST DAMAGE

Studies in Europe and the United States

Intensive research on possible effects of acidic deposition (and its precursors) on forests have been carried out over the last two to three decades both in Europe (UN/EC 2004) and the United States (NAPAP 1998; Driscoll 2001). Menz and

Seip (2004) give a short overview. Nonetheless, quantitative relationships between primary pollutants and forest damage have been difficult to obtain. In Europe, assessment and monitoring of effects of air pollution on forests have been carried out in a joint UN-EC program since the late 1980s (UN/EC 2004). Except for some areas in Eastern Europe, where direct effects of SO_2 probably have played an important role in causing forest defoliation, there are no clear long-term trends that can be related to acidic deposition. Fortunately, the dramatic forest dieback feared by some scientists in the 1980s never materialized. Recent improvements in tree vitality in some areas in Eastern Europe—for example, in Poland—have been partly ascribed to decreased pollution. To date, investigations of possible effects of acidic deposition on forests in the northeastern United States and in Canada have focused on red spruce and sugar maple. There is evidence that acidic deposition has caused dieback of red spruce by decreasing their tolerance to cold (Driscoll et al. 2001). Damage to sugar maples may in some localized areas be caused, at least partially, by loss of base cations (Ca^{2+}, Mg^{2+}) from the soil.

In spite of considerable defoliation in some areas, European forests grow well. The European report (UN-EC, 2004) states: "Forest growth has increased across Europe. This means that today in general both healthy and defoliated trees show larger increments. The absolute growth level of the defoliated trees is, however, lower. Under certain stand and site conditions, nitrogen deposition can contribute to this growth change, but also increasing temperature and carbon dioxide concentration can have stimulating effects. It has to be clarified whether this increased forest growth leads to improved forest condition and functioning in the long term."

Studies in China

China started its research on the impact of acid rain on forests during the 1980s and 1990s, conducting quantitative research on the impacts in 11 provinces (Feng Zongwei et al. 1999; Chen Chuying et al. 1993; Cao Hongfa et al. 1993). These studies revealed that acid rain and SO_2 had obvious impacts on coniferous forest (mainly masson pine and Chinese fir forest). For nine provinces, the volume loss rate attributable to the impact of acid rain and SO_2 on masson pine and Chinese fir was estimated by means of a multi-factor analysis of altitude, slope location, slope, slope orientation, soil thickness, thickness of black soil, SO_2 (average and daily value), and acid rain (average pH) (see table 6.5). These volume loss rates were obtained in specific provinces at certain times and under certain conditions of acid rain and SO_2 pollution, and are not generally applicable. Using these volume loss rates, the forest loss in one province in 2003 was estimated (see Annex).

Uncertainties

In spite of extensive studies, reliable relationships between forest growth and SO_2 concentrations or precipitation pH have not been established in Europe or the United States. In the cost-benefit analysis for the *Protocol to Abate Acidification, Eutrophication, and Ground-Level Ozone in Europe* (Holland et al. 1999), effects on forests (timber production) were only estimated for ozone; the effects of other pollutants were considered too uncertain.

Although exposure-response functions have been suggested in China based on Chinese studies (see Appendix X), they are only tentative. The pH relationships are probably the least reliable. The relationships are given separately for various provinces. This may in some cases reflect differences in environmental conditions, such as soils, but it is far from a satisfactory solution. Furthermore, the studies were carried out more than a decade ago. At least in some regions, there has been an increase in the nitrate/sulfate ratio in precipitation. Tu et al.

TABLE 6.5 Annual Average Timber Stocks Applied in Tree-by-Tree Investigation and Calculated Reduction Rates

	Annual Average Timber Stocks (baseline)(m^3/ha)		Reduction Rates %			
			Masson Pine		Chinese Fir	
Areas	Masson Pine	Chinese Fir	SO_2	Acid Rain	SO_2	Acid Rain
JiangSu	5.75	6.76	8.45	5.20	5.73	4.77
ZheJiang	6.30	10.70	9.88	10.32	8.70	10.70
FuJian	8.49	8.33	3.04	4.86	1.74	5.36
JiangXi	6.30	6.35	4.40	2.60	6.19	5.91
AnHui	4.60	5.60	4.54	10.16	6.44	9.46
HuNan	4.80	5.40	4.01	6.29	6.56	11.74
HuBei	4.40	4.50	9.36	3.54	5.85	8.45
SiChuan*	4.74	3.11	16.68		30.20	
GuiZhou*	5.77	5.07	9.38		14.20	

Source: Feng Zongwei et al., 1999; *Chen Chuying et al., 1993.
Note: *indicates data from 1984–86; others are data from 1992–93.

(2005) found that the ratio between the nitrate and sulfate contributions to acidification in precipitation in the Nanjing area increased from 0.1 in 1992 to 0.3 in 2003. Tang et al. (2004) report significant nitrate concentrations. Since nitrogen is an important nutrient, nitrate deposition may increase growth.

In conclusion, the present basis for estimating forest damage caused by air pollution in China is not satisfactory. One reason is the lack of monitoring data in more remote areas (see section on effects on crops). However, more studies to obtain more reliable exposure-response functions are needed. Acidity (pH) in precipitation and SO_2 concentrations in air are not sufficient to determine possible forest damage, even if good annual averages of these parameters have been obtained at the actual forested sites (as opposed to the present situation, when values are essentially from urban areas). If the equations in Appendix X are used, special consideration must be given to local conditions and the large uncertainties must be emphasized.

Economic Evaluation of Forest Damages

In addition to possible loss of timber caused by SO_2 and acid rain (see Appendix), forest damage entails a number of other effects such as loss of non-timber forest products, carbon sequestration, watershed protection, and recreation. The value of these products is likely to be high. In a case study, Zhang (2001) estimated the total value to be 10 times the timber loss, but with very large uncertainty. Mahapatra and Tewari (2005) found the net present value of non-timber forest products to be four to five times greater than potential timber revenue for two studied sites in India. The recent Millennium Assessment Report (MA 2005) compared results from several countries. In most countries, the marketed value of ecosystems associated with timber and fuelwood production was less than one-third of the total economic value, including non-market values such as carbon sequestration, watershed protection, and recreation.

It is likely that the ratio between the value of the forest regarded as timber and the value of an environmental good varies with the degree of damage.

A small reduction in tree growth rates, say less than 10 percent, may have little or no effect on some non-market values, e.g. soil erosion and recreation.

Although the total value of losses related to forest damage is likely to be several times that of loss of timber, most losses are very difficult to estimate. As a first step, we suggest that loss in CO_2 capture can be calculated and used in combination with the best current value per ton CO_2 for monetization.

6.2

Material Damage

Air pollution causes significant material damage in southern China, where dry sulfur dioxide deposition corrodes or deteriorates a variety of materials, mainly building structures. This chapter, reporting on findings in 14 municipalities and provinces in southern China, estimated the economic cost of this damage to be about 6.7 billion RMB in 2003, with Guangdong, Zhejiang and Jiangsu bearing the highest economic burden and accounting for more than 50 percent of the total damage incurred. While there are a number of uncertainties associated with this cost estimation, if combined with the significant economic burden from crop loss due to acid rain, it is clear that the air pollution issue demands urgent attention.

Air pollution causes material damage by corroding and deteriorating materials. Atmospheric corrosion and deterioration of materials is a cumulative, irreversible process that also takes place in the absence of pollutants. The reactivity to various air pollutants varies greatly between different materials and pollutants. Together with the level of air pollution, particularly SO_2 and O_3, and the pH in precipitation, the deterioration processes also largely depend on meteorological conditions, especially the "time of wetness" (time fraction with relative humidity >80 percent and temperature >0°C) (Kucera and Fritz 1993). Two processes are involved in deterioration of materials. One is corrosion of metals, which are electrochemical processes depending on the presence of humidity. The other is chemical reactions that alter the properties of materials. Materials with basic properties, such as calcium-rich rocks and concrete, may be sensitive to acidic components. Photochemical oxidants such as ozone (O_3) are also capable of damaging certain materials. Economically important materials that are susceptible to ozone damage include elastomers (natural rubber and certain synthetic polymers), textile fibers, and dyes. Culturally important materials, such as a number of artists' pigments and dyes, may also be damaged (U.S. EPA 1996).

CAUSAL AGENTS AND IMPACT PATHWAY

Causal Agents

Previous studies show a relationship between a range of air pollutants and deterioration rates for different materials. The dose-response relationships presented below are generally based on variables such as the concentrations of SO_2 and O_3, the concentration of H^+ in the rain, and moisture. As there are few monitoring data for O_3 in China, we decided to select functions that include only SO_2 and pH in precipitation as variables.

Valuation Scope

The moisture in the air greatly affects the degree to which acid deposition corrodes materials. In the north of China, where the climate is always quite dry and the days with relative wetness greater than 80 percent are infrequent, the damaging effect of dry acid deposition on materials is probably very low. Referring to the final report from the joint Shanxi-Norwegian project *Master Plan Against Air Pollution in Shanxi Province,* the material losses represented only 0.19 percent of the total pollution cost (Shanxi Environmental Information Centre, Norwegian Institute for Air Pollution 1994). Thus, we limit the scope of the valuation to southern China, specifically the provinces or municipalities of Shanghai, Jiangsu, Zhejiang, Fujian, Ahhui, Jiangxi, Guangdong, Guangxi, Hunan, Hubei, Sichuan, Chongqing, Guizhou, and Yunnan.

Valuated Material Types

A wide range of materials is exposed to polluted ambient air, including the materials used in buildings, bicycles, cables, ancient architectural structures, railways, and bridges. Moreover, the types and amount of materials being used are increasing. The numbers of vehicles and air conditioners have increased rapidly in recent years in China. All materials exposed in the acid environment will to some extent be eroded. According to a Chinese study (see Annex A.3) conducted in the 1980s on the effect of acid deposition on materials, the elements that are most damaged are building surfaces and bicycles. Another study (Henriksen et al. 1999: Kai et al. 1999) in Guangzhou also found that the loss of only three materials—outdoor galvanized steel, painted and galvanized guardrails—represented nearly 80 percent of the total material loss. In conclusion, we only include in the valuation the materials with large amounts, extensive distribution area, and the support of exposure-response functions.

With regard to the research finding of the national key scientific and technical project during the seventh “five-year plan period,” buildings and bicycles accounted for 55 percent and 40 percent, respectively, of the total material loss in two provinces of Guangdong and Guangxi in 1985. The valuation model in the design phase of the present work included both building materials and bicycles. During the trial computation period, we found that the proportion of bicycle loss to the total material loss was reduced to only 1.5 percent of the total for Guangdong and 3.4 percent for Sichuan. The reason for this remarkable change may be that both building areas and material prices have grown rapidly, while the amount and price of bicycles increased little in recent years. In addition, although vehicles and air conditioners are under long-term outdoor exposure, they are usually replaced not because their lifetimes have been reduced as a result of air pollution, but due to other reasons. With respect to ancient architectural structures, there is currently no standardized valuation approach. Therefore, building materials are the exclusive valuation object in the following.

EXPOSED MATERIALS

In a valuation at a national level, there are two main methods for estimating exposed stocks: (1) using the indicator of material stocks per capita, or (2) looking at material stocks per unit construction area. While the valuation result from the latter indicator may be more reliable, the former one is more feasible because of deficient updating and comprehensiveness of the statistical data for urban construction areas in China. We apply two different datasets for building material stocks per capita based on the material stocks surveys made in Jinan (see Annex A.3), Shanxi, and the previous study in Guangzhou (Henriksen et al. 1999), as shown in Table 6.6, to calculate the material stocks of a) southeastern and b) southern cities in the southern acid rain region.

The total area of exposed building materials is calculated from the exposed material stocks per capita in m^2/person from Table 6.6 times the population in all cities in the provinces that are included.

DOSE-RESPONSE FUNCTIONS AND MATERIAL COSTS

Several studies on the relationship between materials damage and air pollution in Western countries and China have provided a relatively robust basis for exposure-response functions for a wide range of materials, such as the ECE-ICP program (Kucera et al. 1995; Zhang et al. 1993; Wang et al. 1990). Functions for deterioration rates of specific materials have been derived experimentally, in laboratory or under field conditions. From these functions we derived functions for the relationship between air pollution exposure and the service lifetime. By estimating maintenance and replacement cost related to change in service time, the economic damage can be estimated (Kucera and Fitz 1993; Kucera et al. 1993). For those materials where a quantitative assessment of deterioration rate is difficult to obtain, inspections of physical damage in the field have been used to directly estimate the relationship between air pollution exposure and need for maintenance and replacement, i.e. service lifetime (Kucera and Fitz 1993). In the following, we mainly draw upon two reports: one from China, and a review of dose-response functions from Europe (ECON, 2000).

Exposure-Response Functions from China

During the seventh five-year plan period, one of the key scientific and technological projects was research on acid rain. A working group was organized to conduct both indoor and field-exposure experiments on corrosion of different materials.[1] The field exposure experiments were conducted in Liuzhou (a heavily polluted acid rain area), Nanning (a lightly polluted acid rain area), and Guangzhou.

The recommended dose-response relations—based mainly on indoor tests, but somewhat revised by field experiments—are presented in Table 6.7.

TABLE 6.6 Building Material Stocks Per Capita for Eastern Cities and Other Southern Cities (m^2/ per capita)

Materials	Southeastern Cities	Other Southern Cities
Cement	7.25	18.34
Brick	18.51	13.15
Aluminum	10.03	3.2
Painted wood	1.24	0.56
Marble/granite	9.14	0.47
Ceramics/Mosaic	40.97	7.76
Terrazzo /Cement	22.51	15.17
Painted plaster	18.08	20.59
Tile	2.36	3.28
Galvanized steel	0.29	—
Painted steel	6.69	0.28
Painted steel as guardrail	13.82	13.82
Galvanized steel as guardrail	9.21	9.21

Source: Authors Calculations.

Exposure-Response Functions from Europe

ECON (2000) presented exposure-response functions derived from European studies, including the study by Kucera et al. (1995). These functions were applied to estimate material loss in Guangzhou city by Tian et al. (1999). In order to facilitate the valuation of physical damage, Tian et al. adjusted the functions to better represent the Chinese situation and transferred them into service life-year reduction functions. Table 6.10 renders the functions applied by Tian et al.

Exposure-Response Functions Applied in the ECM

Since the material corrosion functions provided by the Chinese project are based on the practical tests and field experiments in southern China and consistent with the Chinese situation, we decided to apply the functions suggested by the Chinese

TABLE 6.7 Exposure-Response Functions of Materials from the Study of the Chinese Acid Rain Project of the Seventh FYP

Materials	Exposure-Response Functions
Paint	$Y = 5.61 + 2.84[SO_2] + 0.74 \times 10^{4}[H^+]$
Marble	$Y = 14.53 + 23.81[SO_2] + 3.80 \times 10^{4}[H^+]$
Galvanized steel	$Y = 0.43 + 4.47[SO_2] + 0.95 \times 10^{4}[H^+]$
Q235 steel	$Y = 39.28 + 81.41[SO_2] + 21.2 \times 10^{4}[H^+]$
Aluminum	$Y = 0.14 + 0.98[SO_2] + 0.04 \times 10^{4}[H^+]$

Source: Wang Wenxing, Zhang Wanhua et al., 1990
Note: *Y* is the corrosion rate of material in a polluted area, μ*m*/ year; [SO_2] is the ambient concentration of SO_2, mg/m^3; and [H^+] is the H^+ concentration of rain, mol/l.

study in the valuation model. For materials like concrete and bricks that Chinese studies did not take into account, we apply the functions provided by ECON (2000) and Tian et al. (1999). All exposure-response functions and associated parameters used in the final valuation model are summarized in Tables 6.9 and Table 6.10.

ECONOMIC COST ESTIMATION

Table 6.11 presents the exposed building materials of all provinces based on the building material stocks per capita in Table and population of all cities in each province.

Using equation 1 and the building material stocks of each province (Table 6.11) and the monitoring data for air pollution, we get the economic cost of each province in Table 6.12.

The economic cost of material damage from acid rain of all provinces in the southern acid rain region reached about 6.7 billion yuan in 2003. Of all provinces in the southern acid rain region, Guangdong had the highest material cost of about 1.6 billion yuan, followed by Zhejiang and Jiangsu. Their material costs are both about 1.1 billion yuan.

UNCERTAINTIES

1) There are great uncertainties in materials inventory. One reason is that the types of

TABLE 6.8 Exposure-Response Functions of Materials Based on European Studies

Materials	Exposure-Response Functions
Concrete	If $SO_2 < 15\ \mu g/m^3$, L = 50 years, else 40 years
Bricks	If $SO_2 < 15\ \mu g/m^3$, L = 70 years, else 65 years
Bricks with plaster	$L = 1000/(0.124 \cdot SO_2 + 15.5 + 0.013 \cdot Rain \cdot H^+)$
Painted wood	$L = 1000/(1.03 \cdot SO_2 + 87.5 + 0.26 \cdot Rain \cdot H^+)$
Marble	$L = 10000/(103.52 + 0.302 \cdot SO_2 + 0.00487 \cdot Rain \cdot H^+)$
Ceramics and mosaic	If $SO_2 < 15\ \mu g/m^3$, L = 70 years, else 65 years
Concrete with stone grain	If $SO_2 < 15\ \mu g/m^3$, L = 50 years, else 40 years
Paint for outer wall	$L = 1000/(0.28 \cdot SO_2 + 18.8 + 0.07 \cdot Rain \cdot H^+)$
Tiles	If $SO_2 < 15\ \mu g/m^3$, L = 45 years, else 40 years
Galvanized steel as guardrail	$L = 30/(0.51 + 0.0015 \cdot TOW \cdot SO_2 \cdot O_3 + 0.0028 \cdot Rain \cdot H^+)$
Painted steel as guardrail	$L = 1000/(1.37 \cdot SO_2 + 103 + 0.35 \cdot Rain \cdot H^+)$
Zinc	$ML = T^{0.92} \cdot (1.2[SO2]^{0.34} exp(0.011RH + 0.062T - 0.9) + 0.21Rain[H^+]$, $T \leq 10°C$ $ML = T^{0.92} \cdot (1.2[SO2]^{0.34} exp(0.011RH - 0.028T - 0.9) + 0.21Rain[H^+]$, $T > 10°C$

Sources: ECON, 2000; Tian et al., 1999.
Note: *L* is the life expectancy in years; *Rain* is the annual rainfall in mm; *H^+* is the H^+ concentration of rain in mol/l; *TOW* is the fraction of time when relative humidity exceeds 80 percent and temperature is greater than 0°C; and [*SO_2*] is the ambient concentration of SO_2 in μg/m^3.

TABLE 6.9 Exposure-Response Functions for Material Loss Valuation

Materials	Y (μm/year) or L(year)	Literature
Cement	If SO_2 < 15 μg/m^3, L = 50 years, else 40 years	11, 13
Brick	If SO_2 < 15 μg/m^3, L = 70 years, else 65 years	11, 13
Aluminum	$Y = 0.14 + 0.98[SO_2] + 0.04 \times 10\ 4[H^+]$	8
Painted wood	$Y = 5.61 + 2.84[SO_2] + 0.74 \times 10\ 4[H^+]$	8
Marble/granite	$Y = 14.53 + 23.81[SO_2] + 3.80 \times 10\ 4[H^+]$	8
Ceramics/Mosaic	If SO_2 < 15 μg/m^3, L = 70 years, else 65 years	11, 13
Terrazzo/Cement	If SO_2 < 15 μg/m^3, L = 50 years, else 40 years	11, 13
Painted plaster	$Y = 5.61 + 2.84[SO_2] + 0.74 \times 10\ 4[H^+]$	8
Tile	If SO_2 < 15 μg/m^3, L = 45 years, else 40 years	11, 13
Galvanized steel	$Y = 0.43 + 4.47[SO_2] + 0.95 \times 10\ 4[H^+]$	8
Painted steel	$Y = 5.61 + 2.84[SO_2] + 0.74 \times 10\ 4[H^+]$	8
Painted steel as guardrail	$Y = 5.61 + 2.84[SO_2] + 0.74 \times 10\ 4[H^+]$	8
Galvanized steel as guardrail	$Y = 0.43 + 4.47[SO_2] + 0.95 \times 10\ 4[H^+]$	8

Source: Authors Calculation.
Note: *Y* is the corrosion rate of material in a polluted area, μm/ year; *L* is the life expectancy in years; $[SO_2]$ is the ambient concentration of SO_2, mg/m^3; and $[H^+]$ is the H^+ concentration of rain, mol/l.

TABLE 6.10 Parameters in the Valuation Model of Material Loss

Materials	CDL (1)	Y_0 μm/Year (2)	L_0 Year (3)	Y μm/Year (4)	*L* Year (5)	*P* Yuan/m^2
Cement			50		40	22
Brick			70		65	65
Aluminum	10.0	0.141	(1)/(2)	Table 3-3-4	(1)/(4)	200
Painted wood	13	5.63	(1)/(2)	Table 3-3-4	(1)/(4)	20
Marble/granite	160	14.63	(1)/(2)	Table 3-3-4	(1)/(4)	200
Ceramics/Mosaic			70		65	48
Terrazzo/Cement			50		40	26
Painted plaster	13	5.63	(1)/(2)	Table 3-3-4	(1)/(4)	15
Tile			45		40	8
Galvanized steel	7.3	0.45	(1)/(2)	Table 3-3-4	(1)/(4)	16
Painted steel	13	5.63	(1)/(2)	Table 3-3-4	(1)/(4)	16
Painted steel as guardrail	13	5.63	(1)/(2)	Table 3-3-4	(1)/(4)	16
Galvanized steel as guardrail	7.3	0.45	(1)/(2)	Table 3-3-4	(1)/(4)	16

Source: Authors Calculations.
Note: *CDL* is the critical damage limit of material, μm; Y_0 is the corrosion rate of material in clean area, μm/year; *Y* is the corrosion rate of material in polluted area, μm/ year; L_0 is the life expectancy of material in clean area, year; *L* is the life expectancy of material *i* in polluted area, year; *P* is the unit price of a single maintenance or replacement operation, yuan/m^2.

BOX 6.1 Estimating the Cost of Corrosion and Deterioration of Building Materials

The economic cost of corrosion and deterioration of building materials (in yuan/year) is calculated as:

$$C' = \left(1/L - 1/L_0\right) \times P \times S \qquad (1)$$

where L_0 is the life expectancy of the material in clean areas (year); *L* is the life expectancy of the material in polluted area (year); *P* is the unit price of a single maintenance or replacement operation (yuan/m^2), and *S* is the stock at risk (m^2).

TABLE 6.11 Exposed Building Material Stocks of All Provinces in the Southern Acid Rain Region (10,000 m^2)

Materials	Jiangsu	Shanghai	Zhejiang	Fujian	Guangdong	Guangxi	Anhui	Jiangxi	Hubei	Hunan	Sichuan	Chongqing	Guizhou	Yunnan
Cement	19,256	9,264	12,599	5,909	30,036	16,703	22,163	12,822	40,414	19,339	24,484	16,288	9,146	11,853
Brick	49,178	23,658	32,176	15,091	76,709	11,976	15,891	9,193	28,977	13,867	17,556	11,679	6,558	8,499
Aluminium	26,650	12,821	17,436	8,178	41,570	2,914	3,867	2,237	7,051	3,374	4,272	2,842	1,596	2,068
Painted wood	3,295	1,585	2,156	1,011	5,140	510	677	392	1,234	591	748	497	279	362
Marble/ granite	24,286	11,683	15,889	7,452	37,881	428	568	329	1,036	496	627	417	234	304
Ceramics/ Mosaic	108,870	52,374	71,230	33,407	169,816	7,067	9,378	5,425	17,100	8,183	10,360	6,892	3,870	5,015
Terrazzo/ Cement	59,801	28,768	39,126	18,350	93,278	13,812	18,326	10,602	33,417	15,991	20,246	13,468	7,562	9,801
Painted plaster	48,040	23,110	31,431	14,741	74,933	18,748	24,876	14,391	45,361	21,707	27,481	18,282	10,265	13,304
Tile	6,258	3,010	4,094	1,920	9,761	2,987	3,964	2,293	7,228	3,459	4,379	2,913	1,636	2,120
Galvanized steel	777	374	508	238	1,211	0	0	0	0	0	0	0	0	0
Painted steel	17,785	8,556	11,636	5,457	27,741	255	338	196	617	295	374	249	140	181
Painted steel as guardrail	36,727	17,668	24,029	11,270	57,287	12,587	16,701	9,662	30,454	14,573	18,450	12,274	6,892	8,932
Galvanized steel as guardrail	24,484	11,779	16,019	7,513	38,191	8,388	11,130	6,439	20,295	9,712	12,296	8,180	4,593	5,952

Source: Authors Calculations.

TABLE 6.12 Material Loss of All Provinces in the Southern Acid Rain Region (10,000 yuan)

Provinces	Material Losses
Jiangsu	104,882
Shanghai	54,841
Zhejiang	105,104
Fujian	22,037
Guangdong	158,266
Guangxi	15,246
Anhui	11,477
Jiangxi	18,246
Hubei	46,749
Hunan	38,509
Sichuan	36,240
Chongqing	35,985
Guizhou	16,497
Yunnan	10,327
Total	674,407

Source: Authors Calculations.

buildings and materials used vary considerably depending on the economic level and the area. We have applied data from surveys in Jinan, Taiyuan, and Guangzhou, and the differences in material stocks per capita among the three cities are substantial. The average material stocks per capita based on these three cities obviously cannot represent the national level. This implies that the urban material stocks of different scales and economic levels are still the critical issues for the valuation of material loss. More data on the kinds and stocks of exposed material in Chinese cities is needed.

2) The dose-response coefficients applied in the estimation are mainly from Chinese studies of the 1980s. At that time, the main contributor to acid rain in China was sulfuric acid, while now it is beginning to change due to increasing emissions of NO_x in recent years. The content of nitrate in the rain will increase and may imply different effects on materials. The application of the dose-response functions derived 20 years ago for the current situation implies uncertainties.

3) Since there are no monitoring data for some small cities, the cost valuation of these cities is based on monitoring data from neighboring cities. Generally, the air quality of small cities is better than that in big cities, thus the total effect may be overestimated.

Endnote

1. In the indoor simulation tests, the variable factors were the acidity of precipitation and the concentration of SO_2. Other factors were kept constant at the following values: temperature 25°C, relative humidity 80 percent, velocity of wind 0.6 m/s, concentration of O_3 20 ppb, exposure time 500 hours. The total exposure period was separated into 42 cycles of 12 hours: rain for 0.5 hours, light for 4 hours, moisture in the form of dew for 3.5 hours, and light again for 4 hours. Thus total exposure to light was 336 hours, to rain 21 hours, and to dew 143 hours.

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