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Coal Use in the People's Republic of China, Volume 1: Environmental Impacts

by N. Bhatti, M.M. Tompkins, J.L. Carlson,* and D.R. Simbeck*

Decision and Information Sciences Division,
Argonne National Laboratory, 9700 South Cass Avenue, Argonne, Illinois 60439

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* *Carlson is affiliated with Illinois State University, Normal, Illinois. Simbeck is affiliated with SFA Pacific, Inc., Mountain View, California.*

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NOTATION

The following is a list of the acronyms, initialisms, and abbreviations (including chemical symbols and units of measurement) used in this document.

ACRONYMS, INITIALISMS, AND ABBREVIATIONS

AFBC	atmospheric fluidized-bed combustion
EPB	Environmental Protection Bureau
EPC	Environmental Protection Commission
FERC	Federal Energy Regulatory Commission
GDP	gross domestic product
IEA	International Energy Agency
IGCC	integrated coal gasification combined-cycle
NEPA	National Environmental Protection Agency (China)
PC-fired	pulverized coal fired
TSP	total suspended particulate
VOC	volatile organic compound

CHEMICAL SYMBOLS AND ABBREVIATIONS

CH_4	methane
CO	carbon monoxide
CO_2	carbon dioxide
NO_2	nitrous oxides
NO_x	nitrogen oxides
SO_2	sulfur dioxide

UNITS OF MEASUREMENT

kWh	kilowatt-hour
$lb/10^6$ Btu	pound(s) per million British thermal units
mtce	metric tons coal equivalent
$\mu g/m^3$	microgram(s) per cubic meter
ppm	part(s) per million
10^6 t/yr	millions of metric tons per year
t	metric ton(s)
wt%	weight percent

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ABSTRACT

The People's Republic of China (hereafter referred to as China) is the largest producer and consumer of coal in the world. Coal makes up 76% and 74% of China's primary energy consumption and production, respectively. This heavy dependence on coal has come at a high price for China, accounting for a large share of its environmental problems.

This report examines the dominance of coal in China's energy balance, its impact on the environment, and the need for technical and financial assistance, specifically for two distinct aspects: the effect of coal use on the environment and the importance of coal to China's economy. The results of the analysis are presented in two volumes. Volume 1 focuses on full fuel cycle coal emissions and the environmental effects of coal consumption. Volume 2 provides a detailed analysis by sector of China's economy and examines the economic impact of constraints on coal use.

1 INTRODUCTION

Since the formation of the People's Republic of China (hereafter referred to as China) in 1949, that country has experienced rapid economic and social development. In the past four decades, China's population has increased by 602 million, which represents an annual growth rate of 1.9%. Although family planning measures instituted in the early 1970s have slowed population growth, China's population reached 1.1 billion in 1990, the highest population of any country in the world. The population is mainly rural and overwhelmingly located in the central and southern provinces (only 26% lived in urban areas in 1990) (Sinton 1992). Economic reforms in the late 1970s opened the domestic economy to the West and accelerated modernization. As a result, urban population has grown faster than rural population, a trend expected to continue in the future.

1.1 OVERVIEW OF THE DOMESTIC ECONOMY

Providing basic human services for such an enormous population remains one of China's major economic problems. While changes in the nation's economic system have promoted rapid economic growth, an average growth in gross domestic product (GDP) of 9% during the 1980s has resulted in a low per capita GDP. In 1990, the per capita GDP was less than $350 per year; this figure was calculated by using the average official exchange rate of 4.78 yuan per dollar (Sinton 1992). In the last decade, China's economic reform program has improved incentives and productivity by decentralizing decision making, giving more autonomy to businesses, and opening the domestic economy to world markets.

However, China's economy succumbed to the mounting inflationary pressure of rapid expansion in the early 1990s. In the last two years, austerity measures have stabilized inflation, but little progress has been made to ensure the financial accountability of state enterprises, which is one of the underlying causes of inflation. Price reform is widely recognized as an essential part of financial accountability, but price reform has been slow, particularly in the energy sector. To date, the energy sector is characterized by a multitiered pricing system: the high state price, the low state price, the market price, and the international price. While this multitiered pricing system has effectively increased production because it allows state enterprises to sell above plan output at market prices, this system has its drawbacks. For example, it has contributed to the inefficiency of state enterprises and has exacerbated fraudulent business practices. Although prices have been adjusted upward in state enterprises (e.g., freight, coal, and gas) in the last two years, state-controlled prices for energy and transport are still significantly below market-determined prices.

The key distinguishing feature of China's economy is that its industrial sector dominates the GDP. Industry and manufacturing accounted for 56% of China's GDP in 1990, which makes China one of the world's leading producers of industrial products. While industry's share of China's total output is high when compared with that of other developing countries, it is also high when compared with that of industrialized nations. The sectoral shares of agriculture and services were approximately equal in 1990; that is, each accounted for 22% of China's GDP. Both manufacturing and services grew rapidly during the 1980s, but at the expense of agriculture. The composition of China's domestic economy is expected to continue on its current development path; that is, the share of agriculture is expected to decline, and the shares of industry and services are expected to increase in coming decades.

1.2 ENERGY DEMAND AND SUPPLY

China is one of the largest consumers of energy in the world. Economic growth and reform have significantly changed China's energy sector. International trade of primary energy sources makes up a small portion of China's energy balance; therefore, energy consumption closely follows economic activity. It is not surprising then that China's largest energy consumer is the industrial sector. Industry accounted for approximately 68% of total end-use energy in 1990. Chinese households use less energy, particularly from high-grade sources such as electricity, than those of other developing countries. Because China's population is mostly rural, with little access to commercial energy sources, biomass dominates residential energy consumption. Although coal is still the major commercial energy source for all sectors except transportation, the proportion of electricity has increased. Table 1 lists China's commercial end-use energy consumption and its sectoral composition in 1990.

TABLE 1 Commercial Energy End-Use Consumption and Composition in China, 1990

Sector	Consumption (mtce)[a]	Share (%)
Agriculture	48.5	4.9
Industry	687.9	69.7
Transportation	45.4	4.6
Service	47.2	4.8
Residential	158.0	16.0
Total	987.0	100.0

[a] mtce = metric tons coal equivalent.

Source: Ministry of Energy (1992).

The energy intensity of China's economy (as measured by primary energy consumption divided by GDP) has decreased steadily since 1977 but remains high relative to other countries. For example, China's energy intensity is approximately three times that of industrialized countries and twice that of developing countries. This discrepancy in energy intensities is largely attributable to China's low technical efficiency, subsidized energy prices, and the large share of industrial products in the composition of the GDP. In addition, energy-intensive industries such as steel and cement manufacturing account for about 37% of industrial energy consumption. These figures reflect the priority given to development of China's heavy industry before 1981.

Faced with energy shortages and increasing evidence of energy inefficiencies, China implemented a comprehensive energy conservation program in 1981. Energy-efficient technologies, modernization of capital equipment, and the shift in industrial production to light industry and services helped to reduce China's energy intensity by about 30% during the 1980s. Energy consumption and production grew by only 5% during the 1980s, that is, less than 50% of the rate of growth in GDP over the same period. Energy planners are promoting additional reforms to improve economic and energy efficiency, while minimizing damage to the environment. Elements of the current reform program include increased imports of energy-efficient and environmental technologies, continued dissolution of monopoly energy corporations, increased energy prices to cover costs and discourage inefficiency, and elimination of subsidies to state enterprises.

China is the world's largest producer and consumer of coal. Production in 1990 exceeded 1 billion metric tons, which accounted for approximately 30% of the world's coal supply. Coal, which is the dominant fuel in China's energy balance, makes up 76% of the nation's primary energy consumption and 74% of its primary energy production. Petroleum began contributing a significant share of total primary energy production and consumption in the mid-1960s, as petroleum deposits were discovered in northeastern China. Coal has since regained its market share in primary energy consumption, as increasing quantities of coal were exported and the growth in oil production decelerated. As shown in Table 2, oil currently accounts for approximately 17% of China's primary energy consumption and production. Although historically China has relied primarily on domestic energy sources, it may become a net importer of oil early in the next century to relieve its projected energy shortage.

Although coal is the largest contributor to China's energy supply, approximately 80% of the energy used by rural households (800 million people) is composed of traditional biomass fuels (e.g., firewood, crop by-products, and animal waste). Most of China's energy-related environmental problems are caused by its dependence on coal and heavy use of biomass. Environmental pollution will remain a major problem and challenge for China's energy industry for the foreseeable future.

1.3 INTERNATIONAL TRADE

The aftermath of the political crisis of 1989, with the suspension of Western loan and aid programs, made it even more essential for China to limit unnecessary expenditure of foreign reserves and to increase exports to rebuild these reserves. This policy accounts for part of the 22% decrease in U.S. exports to China and the simultaneous 27% increase in U.S. imports from China in the early 1990s. In 1991, China's trade surplus with the United States exceeded $12 billion, up from $2.8 billion in 1987. Government policies to restrict

TABLE 2 Primary Commercial Energy Production and Composition in China, 1990

Sector	Consumption (mtce)[a]	Share (%)	Production (mtce)	Share (%)
Raw coal	752	6.2	729	75.3
Crude oil	164	16.6	168	17.3
Natural gas	20	2.0	20	2.1
Hydroelectric	51	5.2	51	5.3
Total	987	100.0	968	100.0

[a] mtce = metric tons coal equivalent.

Source: Asian Development Bank (1991, 1994).

imports and encourage growth in export-oriented industries contributed to China's favorable trade balance (Conable and Hampton 1992/93).

Recognizing the importance of foreign direct investment, China also implemented policies such as tax holidays, tariff exemptions, and copyright laws to improve the investment climate and encourage technology transfer. The result has been a dramatic increase in foreign direct investment, particularly from Hong Kong and Taiwan. Growing interdependence of China's economy with Hong Kong and Taiwan, collectively known as Greater China, contributed to the strong export growth experienced during the last few years. For example, Hong Kong has shifted much of its manufacturing and assembly operations to China in order to take advantage of lower labor and land costs. In contrast, Japan and other developed countries have not played a significant role in the Chinese economy through foreign investments.

The issue of the availability of foreign reserves is critical for China because hard currency cannot be purchased in foreign exchange markets (nonconvertible currency). China has, therefore, only three routes to acquire foreign exchange: exports, foreign direct investment, and borrowing. China's economic reform program has emphasized both increasing exports and foreign direct investment. Consumer imports are tightly controlled, making the demand for foreign exchange largely a function of China's modernization program.

In terms of product, China's trade is rather diversified: manufactured goods account for two-thirds of products, and primary products (including energy) account for the remainder. Despite the infusion of foreign technology and the concomitant use of hard currency, the energy sector's trade account has historically run a surplus of several billion dollars. Oil dominates China's energy trade, and coal plays only a minor role.

1.4 ENVIRONMENTAL POLICY

As with many centrally planned economies, the environmental implication of industrial development was not given a high priority during the early growth stages. Finally, in 1979, China passed the Law of Environmental Protection and established the Environmental Protection Commission (EPC). The EPC's executive body, the National Environmental Protection Agency (NEPA), established in 1988, together with the provincial Environmental Protection Bureaus (EPBs) formulate and implement environmental regulations. Similar to the U.S. environmental protection system, China's NEPA sets policy, priorities, and standards that are then enforced by the provincial bureaus. As China's environmental enforcement agents, EPBs monitor the discharge of pollutants into the air and water, levy discharge fees on local enterprises, and conduct environmental impact assessments for local projects. The Environmental Protection Law of 1989, which further clarified the responsibilities of NEPA and the EPBs, together with existing regulations on marine, air, and water, provides China with a comprehensive, legal framework for environmental protection that compares favorably with other developing countries.

The National Environmental Protection Agency's basic principles follow international standards such as "polluter pays" and "prevention first." For example, under prevention first, China requires medium- and large-scale construction projects to establish a system for addressing environmental impacts with a minimum of 7% of their total investment. Following the polluter pays strategy, NEPA assesses discharge fees and punitive fines, and the EPBs collect for pollution discharge.

While the legal structure for environmental protection exists, significant obstacles preclude successful implementation of environmental regulations. A major deterrent has been the lack of public support for NEPA's environmental programs. China's average citizen has little understanding of the health implications of pollution and, having lived through decades of poverty, gives higher priority to economic development than to environmental problems. However, environmental awareness and the demand for health are expected to increase along with China's overall standard of living.

Market-based mechanisms promulgated by NEPA have been limited in their effectiveness. Pollution fees and fines are generally set below the cost of treating the pollution and are, therefore, too low to provide the economic incentive for changing polluting practices. In addition, state-run enterprises, which account for a disproportionate share of China's pollution, often circumvent the pollution fee through an "IOU" system of accounts.

China's pollution problems have been exacerbated by the resource distortions of central planning. Subsidized prices for coal, water, and other raw materials have led to overuse and waste. Environmental problems related to energy use reflect the overall structure of energy supply and demand. Because China relies heavily on coal, particularly in the residential and commercial sectors, the environmental effects of energy consumption are closely related to coal use. One of the main objectives of China's environmental policy, therefore, is to prevent and control air pollution caused by coal combustion and simultaneously achieve greater efficiency in coal use. Toward that goal, China has adopted the following policy measures:

- Reduce reliance on coal by developing hydropower and nuclear energy;
- Substitute electricity for direct burning of coal by expanding coal-based electricity-generating capacity;
- Improve coal-use efficiency: construct district heating facilities, increase the use of briquettes, and increase coal gasification or use of town gas (for cooking);
- Improve industrial boiler technology;
- Increase coal washing;
- Reduce power plant emissions by using dust collectors (for ash) and flue-gas desulfurization equipment for sulfur dioxide (SO_2); and
- Increase the use of coal solid wastes.

Until recently, global environmental issues such as carbon dioxide (CO_2) emissions had little impact on formulating China's environmental and energy policies. In addition to advanced coal treatment techniques (washing and briquettes) and more efficient combustion technologies, significant reductions in CO_2 emissions are likely to limit the volume of coal use. Because coal plays such an important role in China's energy balance, the effects of constraints on coal use will be manifested throughout the domestic economy (Haugland and Roland 1990).

1.5 PURPOSE OF STUDY

The dominance of coal in China's energy balance, its impact on the environment, and the need for technical and financial assistance, have motivated the current research effort. This study examines two distinct aspects of coal use in China: the effect of coal use on the environment and the importance of coal to the Chinese economy. The results of the analysis are presented in two volumes. Volume 1 focuses on full fuel cycle coal emissions and environmental effects; Volume 2 provides a detailed analysis by sector of the Chinese economy and examines the economic impact of constraints on coal use (Tompkins et al. 1994).

Assessing the environmental impacts of coal use requires a comprehensive understanding of the total energy requirements and pollutants attributable to coal use, that is, the full coal fuel cycle. Because emissions data are not available for the full coal fuel cycle, a spreadsheet model was developed to estimate the emissions associated with the various stages. Discharges into all media were estimated, although very little quantitative information was available for soil and water. Air emissions were calculated on the basis of the quantity of coal consumed, the specific coal characteristics, and the technology being used. On the basis of the estimated discharges, observed and potential impacts of coal use on the different environmental media were assessed. Consumption activities were emphasized because the most serious environmental disruptions occur as a result of combustion, followed by extraction.

Unfortunately, because the adverse impacts to air, water, and land resources are not viewed as a cost of doing business, these effects are not factored into the decision-making process. Translating the environmental effects of coal use into an economic context involves estimating the full costs and benefits from the societal perspective. Theoretically, once the environmental effects have been identified and the external costs estimated, it is possible to determine the optimal level of coal consumption. Because of the paucity of quantitative information, external costs could not be estimated directly for China. Transferability of externality costs from other countries was examined as a possible solution; however, upon a review of numerous externality cost studies, it was determined that none of the estimates was transferable to the Chinese situation.

Although a full cost-benefit analysis was not possible given the data, the economic effect of environmental constraints on coal use could be determined. In particular, a dynamic linear programming model was developed to assess how various sectors in China's economy would adjust to CO_2 emissions constraints. This model is based on the 1987 Chinese input-output table, which is the most detailed sectoral production information available. Five strategies to reduce CO_2 emissions were assessed: (1) change in sectoral mix, (2) mandated conservation, (3) interfuel substitution with current technology, (4) interfuel substitution with technological advances, and (5) a combination of strategies. The basic scenario examined the economic impacts of achieving a 20% reduction in year 2000 baseline CO_2 emissions by the year 2025. The surprising conclusion of the economic analysis is that China is able to reduce CO_2 emissions substantially by the year 2025 with little or no impact to economic growth, as measured by the GDP.

1.6 REPORT ORGANIZATION

The emissions estimation, analysis of the environmental effects, and discussion of externality costs are presented in this volume. It is divided into four sections that correspond to the areas discussed above: introduction, coal fuel cycle analysis, environmental impacts from continued reliance on coal, and an overview and assessment of the transferability of available externality cost estimates to China.

The economic analysis is contained in *Coal Use in the People's Republic of China, Volume 2: The Economic Effects of Constraining Coal Utilization* (Tompkins et al. 1994). The underlying assumptions, model framework, and simulation results are also provided there.

2 COAL FUEL CYCLE ANALYSIS

2.1 COAL RESOURCES AND DEMAND

2.1.1 Coal Resources

Coal-bearing areas within China total 550,000 km^3; China ranks second only to the former Soviet Union in coal reserves. According to the Ministry of Energy, China's proven coal reserves totaled 967 billion tons in 1991. However, the World Energy Conference defines proven reserves as deposits that have actually been explored. These deposits make up only 30% (290 billion tons) of those reported by the Ministry of Energy. On the basis of the World Energy Conference definition, China ranks behind the United States and the former Soviet Union in proven reserves. China's coal reserves are unevenly distributed geographically — most of the country's coal mines are located north of the Qinling-Kunlun mountains. These reserves account for more than 84% of the nation's proven deposits. Given China's vast coal resources, it is unlikely that the availability of coal will constrain the use of coal in the near term (Ministry of Energy 1992).

2.1.2 Coal Demand

China is the most populous country in the world — its 1.1 billion people represent about one-fifth of the world's population. China is also the world's largest coal producer and consumer. The country currently produces and consumes about 1.1 billion tons of coal per year. Coal utilization in China is uniquely different from that in the United States and other major coal-consuming countries — coal represents 75% of China's total energy consumption, whereas in the United States, coal is only 24% of its total energy consumption. China's coal consumption (by sector) is as follows:

- Utility applications: 30%,[1]
- Industrial applications: 50%, and
- Residential and commercial applications: about 20%.

Projections in Figure 1 show that most of the world's growth in coal use for the foreseeable future will occur in China. China's increasing coal consumption, combined with its rapidly increasing standard of living, leads to the largest growth in energy consumption and emissions in the world.

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[1] This amount is drastically different from that in the United States, where utility applications account for 87% of coal consumption.

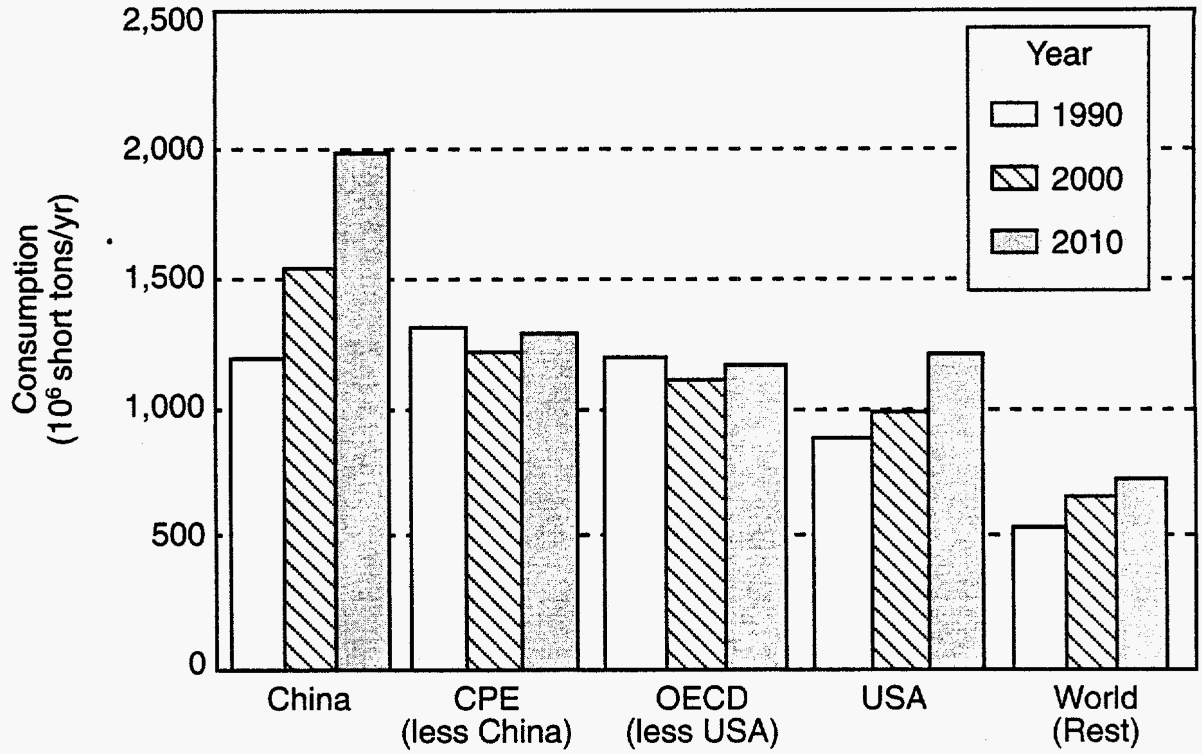

FIGURE 1 World Coal Consumption (Source: Adapted from Simbeck et al. 1993)

2.2 ENERGY AND EMISSIONS FOR COMPLETE COAL FUEL CYCLE

A spreadsheet model has been developed to estimate and document the key emissions generated by coal use in China. Because coal represents about 75% of China's entire energy consumption and produces significantly more emissions per unit of energy consumed than other fossil fuels, this model calculates most of China's energy-related emissions.

The coal fuel cycle is divided into four key stages:

- Extraction,
- Preparation (cleaning),
- Transport, and
- End use:
  - Combustion and
  - Noncombustion.

Figure 2 shows a schematic of China's coal fuel cycle. Combustion and noncombustion end uses are further divided into final consumption sectors. End-use combustion is characterized by rail, residential coal, industrial boilers, and utility boilers. The primary category of end-use noncombustion is coking. The end-use sectors are shown in Figure 3.

The small amount of coal used directly at the mine and during coal preparation is included in these stages of the fuel cycle. End use primarily involves direct coal combustion. Noncombustion end use accounts for about 15% of the coal used, primarily coke ovens (for steelmaking) and coal gasification (for town gas).

The model calculates emission factors and emissions for each stage in the coal fuel cycle for the following pollutants and waste streams:

- Air emissions:
  - Methane (CH_4);
  - CO_2;
  - SO_2;
  - Nitrous oxides (as NO_2);
  - Particulate (fly ash);
- Solid waste; and
- Liquid waste.

The standard method for calculating air emissions depends on the quantity of coal being consumed, the specific characteristics of the coal being burned, and the technology

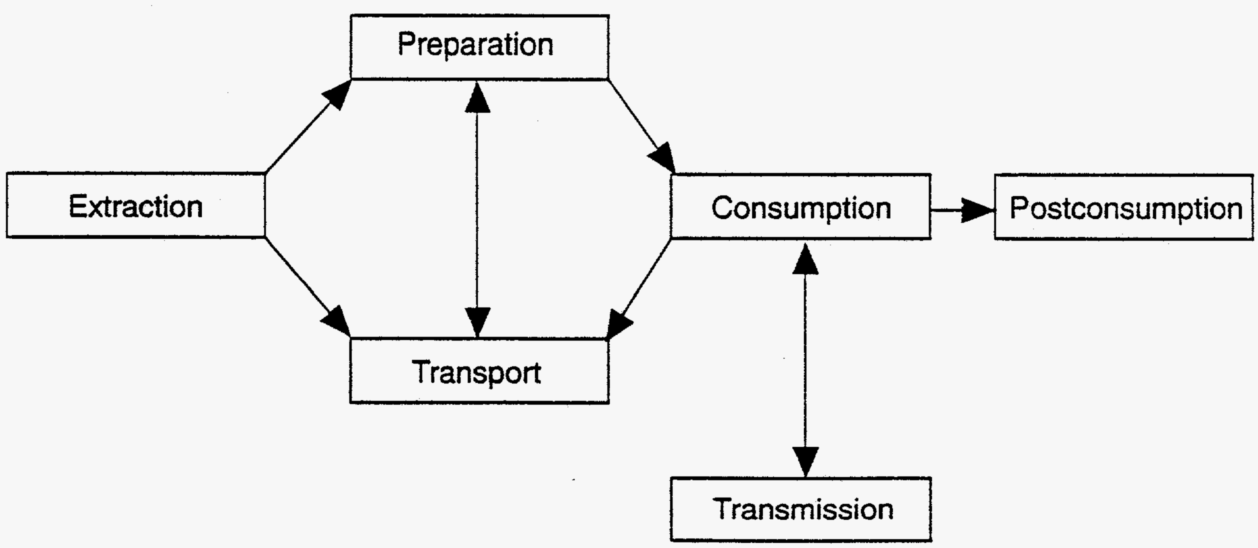

FIGURE 2 Schematic of the Coal Fuel Cycle

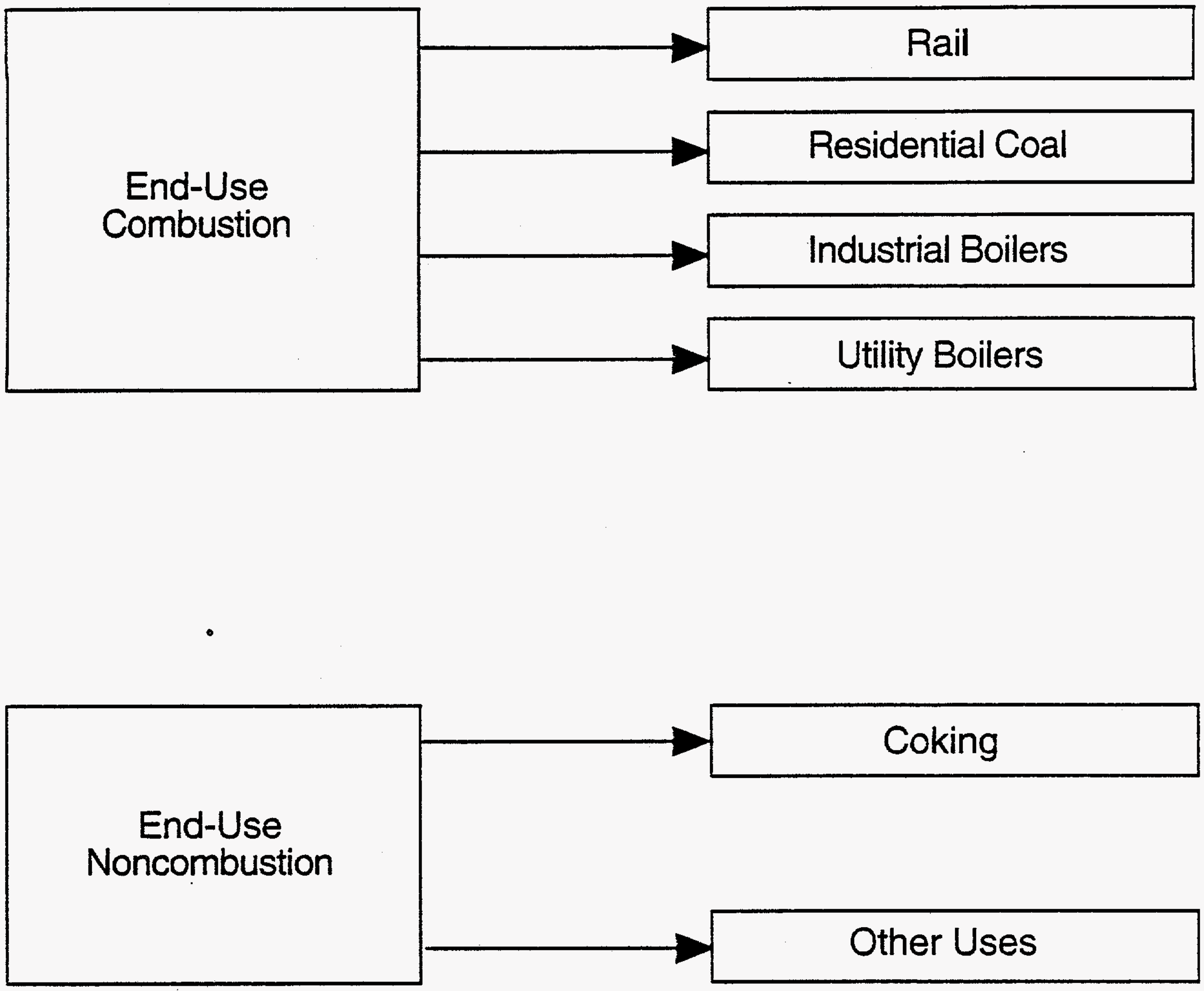

FIGURE 3 Characterization of End-Use Coal Consumption

being used. In general, coal characteristics and the technology are combined into an "emissions factor." This factor represents the amount of a particular emission per unit. Thus, emissions can be calculated according to the following formula:

$$\text{Emissions} = \text{Quantity Coal Consumed} \times \text{Emissions Factor} \times (1 - \text{Recovery Rate}).$$

The derivation of emission factors cannot be summarized generically because the specific method varies by pollutant and media. For example, the SO_2 emissions factor is derived on the basis of the coal's sulfur content, while the NO_x derivation is a function of the combustion process.

Emission factors are based on technical parameters associated with a particular stage in the fuel cycle as well as raw coal characteristics. Key technical parameters used in estimating the emission factors for coal combustion for each stage of the fuel cycle are presented in Table 3.

Essentially all of the air emissions are generated during coal combustion, except for CH_4 emissions, which are produced during coal mining. Solid waste is primarily produced as coal cleaning waste from coal preparation and the larger ash particles that remain after combustion (which do not become fly ash emissions). Liquid waste is primarily produced as a result of removing mine acid drainage water during mining. Estimates of emissions from the coal fuel cycle analysis are shown in Table 4. All emissions are presented in units of millions of metric tons per year. Unit emissions in pounds per million Btu of fuel were also calculated for comparison with other countries emissions and standards. The following sections describe the method, assumptions, and results according to sector.

2.2.1 Extraction

The key assumptions in calculating emissions from coal mining are quantity of raw coal production, type of mining method, and quality of coal. In 1991, China's raw coal production totaled 1,087 million metric tons. Large mechanized state-controlled mines produce about 45% of China's coal. These mines generate large amounts of fines and increase the moisture in the coal because they use water to control coal dust during mining. Locally operated state-owned mines, which account for about 20% of China's overall production, are partially mechanized. Privately owned mines account for the remaining 35%, and these mines exhibit little or no mechanization (Ministry of Energy 1992).

An important aspect of China's coal industry is its geographic distribution. Although most provinces produce some coal (Table 5 and Figure 4), coal production is concentrated in the northern provinces. The 1984 province-level data provided in Table 5 represent the most recent information available on coal production and consumption by province (Doyle 1987).

TABLE 3 Key Technical Parameters Used in Estimating Emissions Factors for Coal Combustion

Sector	Carbon Conversion (wt%)	Sulfur Recovery (wt%)	Ash as Fly Ash (wt%)	Fly Ash Control (wt%)	Stack Height	Nitrogen Conversion (mol%)
Mining	90	—	20	25	Medium	30
Preparation	90	—	20	25	Medium	30
Rail	85	—	15	0	Low	35
Residential	85	12	15	0	Low	35
Industrial boilers	90	1	20	5	Medium	30
Utility boilers	95	5	80	90	High	25

TABLE 4 Coal Fuel Cycle Emissions Summary, 1991 (in millions of metric tons)

				End Use		
Waste Stream	Mining	Preparation	Transport	Combustion	Noncombustion	Total
CH_4	5.3	0.0	0.0	0.4	1.2	7.0
CO_2	18.6	10.1	19.0	1,519.5	372.1	1,939.3
SO_2	0.2	0.1	0.0	15.1	0.4	15.9
NO_2	0.1	0.0	0.3	6.8	1.2	8.3
Particulate/fly ash	0.5	0.3	0.1	22.4	2.8	26.2
Solid waste	7.7	54.1	4.7	191.9	34.6	292.9
Liquid waste	1,445.1	61.5	0.9	409.0	40.8	1,957.3

TABLE 5 Coal Production, Consumption, and Balance by Province in China, 1984 (in millions of metric tons)

Province	Production	Consumption	Balance
Beijing	9.1	23.1	-14.0
Tianjin	0.0	17.3	-17.3
Hebei	64.3	77.6	-13.3
Shanxi	246.5	72.5	174.0
Inner Mongolia	37.4	34	3.4
Liaoning	45.9	79.3	-33.4
Jilin	22.3	36.2	-13.9
Heilongjiang	71.7	55.9	15.8
Shanghai	0.0	24.7	-24.7
Jiangsu	23.3	60.7	-37.4
Zhejiang	1.4	23.4	-22.0
Anhui	30.5	31.6	-1.1
Fujian	8.5	12.1	-3.6
Jiangxi	20.5	23.2	-2.7
Shandong	55.6	64.7	-9.1
Henan	82.5	62.7	19.8
Hubei	10.0	33.7	-23.7
Hunan	35.6	41.1	-5.5
Guangdong	9.3	22.8	-13.5
Guangxi	10.4	15	-4.7
Hainan	0.0	0	0.0
Sichuan	67.1	66.8	0.3
Guizhou	32.1	23.9	8.2
Yunnan	20.6	21	-0.5
Xizang	0.0	0	0.0
Shanxi	27.7	25.5	2.2
Gansu	13.6	17.1	-3.5
Qinghai	2.7	4.5	-1.8
Ningxia	13.3	7.3	6.0
Xinjiang	18.1	16	2.1
Total	979.7	993.7	-14.00

Source: IEA Coal Research (1987).

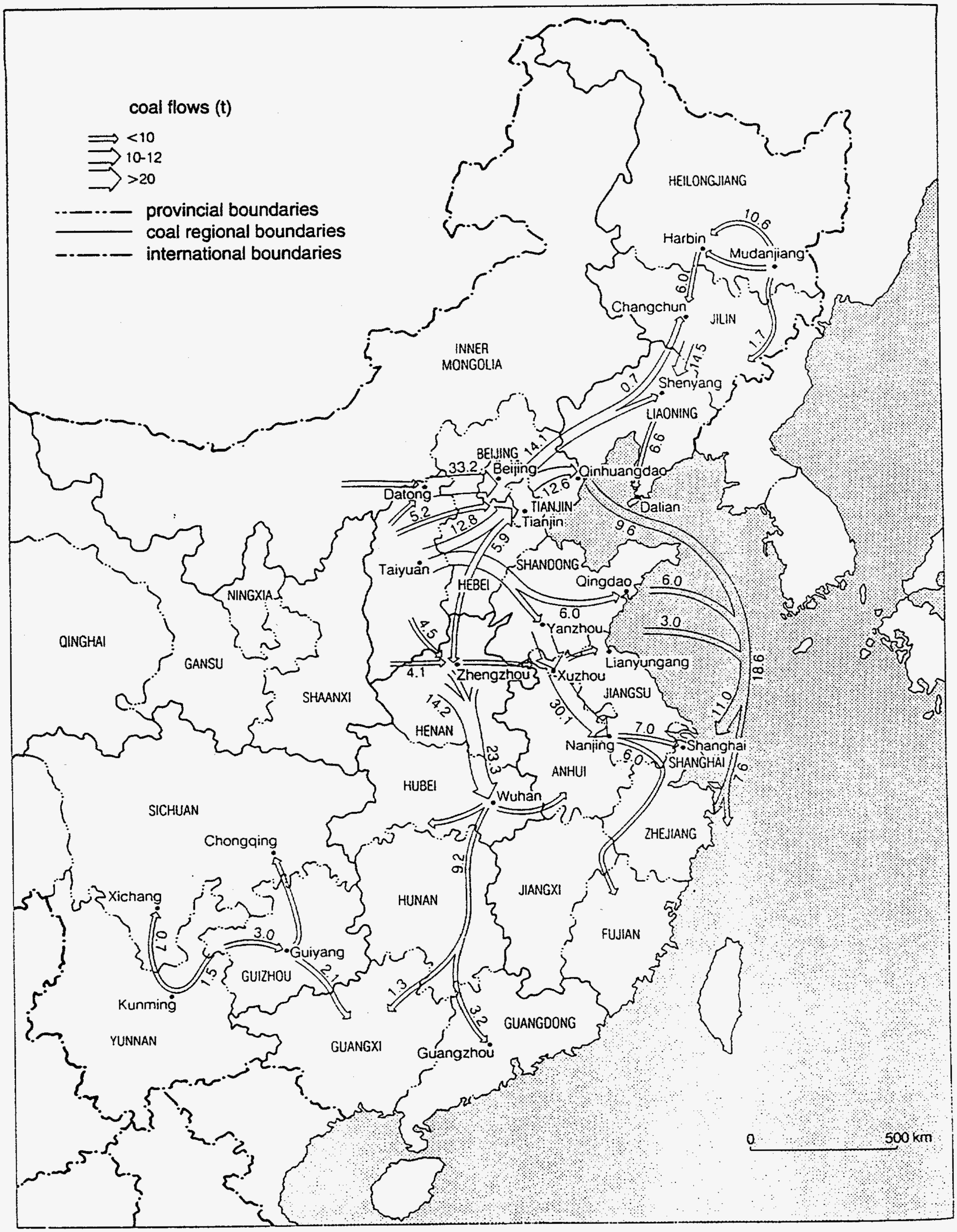

FIGURE 4 Major Coal Flows in China, 1980 (Source: Adapted from Doyle 1987)

Nearly 75% of China's coal is high-rank bituminous, which is low in sulfur; however, China also has moderate amounts (21%) of low-sulfur anthracite coal. High-sulfur coals are found primarily in southern China, whereas most of China's coal is produced in the north central and northeastern regions. The aggregate sulfur content is a key assumption for determining final SO_2 emissions.

The high ash and moisture contents of Chinese coals are largely attributed to the system of remuneration for miners. Payment is based solely on the quantity of coal mined, and there are no controls on energy content. This procedure encourages miners to mine the roof and floor above and below the coal seam to increase the tonnage. It also encourages excessive additions of water. However, China is currently reforming the state-controlled mining and allocated pricing systems, which should eventually resolve this problem.

The principal fuel consumed at the mine is electricity. China's coal mining industry consumes electricity at the rate of 47 kWh/t of coal produced. Other fuels used by mining operations include coal and wood. About 11 million metric tons of coal or 1% of raw coal production is consumed at the mine. This coal is used as boiler fuel for heat and power and is treated as coal consumed above raw production in this analysis. Coal emissions associated with electricity used at the mine are included as part of the end-use sector (Ministry of Energy 1992).

Mining emissions arise from two sources: direct coal combustion at the mine and the mining process. Emissions from direct coal combustion are a function of the type of coal being burned. The key coal characteristics that drive the emissions factors for the various pollutants are listed in Table 6.

The coal characteristics used in the coal analysis model represent a weighted average of the various grades of coal in China. This assumption is important because emissions are

TABLE 6 Raw Coal Characteristics

Characteristic	Moisture and Ash Free	Moisture Free	Run of Mine
Moisture (wt%)	NA[a]	NA	20.00
Ash (wt%)	NA	24.09	19.27
Carbon (wt%)	85.46	64.88	51.90
Hydrogen (wt%)	5.34	4.05	3.24
Oxygen (wt%)	6.40	4.86	3.89
Sulfur (wt%)	1.48	1.13	0.90
Nitrogen (wt%)	1.32	1.00	0.80
Heating value (Btu/lb)	5,306	1,619	9,295
Heating value (10^6 Btu/t)	33.75	25.62	0.50

[a] NA = not applicable.

largely driven by the grade of coal. For example, more than 95% of China's coal is classified as bituminous and anthracite coal. Both are low-sulfur coals that give an average sulfur content of 0.90 wt%. A key parameter of the coal analysis is the assumed heating value for Chinese coal. Because this parameter is critical and affects all emission factors for direct coal combustion, it was chosen to match the reported Chinese heating value. A direct consequence of preserving the consistency in heat values is the assumed moisture content (20% by weight), which is high relative to most published data.

As shown in Table 7, the principal emission generated during mining is liquid waste. The groundwater could be drained before mining and effectively used for irrigation because most coal is mined in semiarid regions that need water. However, once this water enters the mining area, it reacts with the sulfur in the coal and air to make dilute sulfuric acid, commonly called coal mine acid drainage.

Mining also produces a large portion of China's CH_4 emissions. Emissions of CH_4 could be reduced effectively by recovering coal bed CH_4 before the coal is mined. This technique can also be used to simultaneously recover the groundwater.

2.2.2 Preparation

Coal preparation, commonly called coal washing or coal cleaning, involves removing particles with a high ash content from the raw coal in water-based systems. Doing so reduces the ash content of the final coal product. As in the United States, essentially all metallurgical (coking) coals and export steam coals are cleaned in China; most domestic steam coals are not. Only 196 million metric tons or 18% of China's raw coal is washed.

The low level of coal preparation has important implications for the efficiency of the coal sector. As noted, output targets for coal production are based on tons of unwashed raw coal produced. This system leads to the high ash content in run-of-mine coals in

TABLE 7 Total Coal Mining Emissions, 1991 (in millions of metric tons)

Waste Stream	Coal Combustion	Coal Mining	Total
CH_4	0.05	5.33	5.38
CO_2	18.62	0.00	18.62
SO_2	0.20	0.00	0.20
NO_2	0.09	0.00	0.09
Particulate/ fly ash	0.40	0.11	0.51
Solid waste	2.26	5.44	7.70
Liquid waste	5.05	1,440.00	1,445.06

China and in low heating values. The high fines content of Chinese coals increases moisture, coal losses, and coal preparation costs. In addition, nearly 10% of the feed coal energy is lost in the cleaning process, which creates serious problems with solid waste emissions.

The low rate of coal washing also means that most of the coal transported contains relatively large amounts of dirt, rocks, and other inert matter. Transporting materials that are not coal is inefficient and taxes an already overburdened rail system; it also reduces boiler/furnace efficiency and exacerbates the residue disposal and pollution problems. China is aggressively promoting increased coal preparation and plans to increase coal cleaning capacity to 300-350 million metric tons by the year 2000. This additional coal cleaning capacity is expected to increase the availability of high-quality coals for both steam and coking (IEA Coal Research 1992).

In 1991, the raw coal sent to preparation plants totaled 196 million metric tons; however, these plants produced only 82 million metric tons of washed coal (Ministry of Energy 1992). This figure is unreasonably low, given the historical difference of 20 million metric tons between raw coal production and consumption. According to Sinton et al. (1992), China produces large quantities of middlings (small pieces of steam coal) as a by-product of the preparation process. Apparently, China does not include middlings as part of clean coal production, which explains the low reported output. The coal preparation analysis assumes an output of 143 million metric tons on the basis of available consumption and production information of the economy as a whole. This figure is consistent with the sectoral end-use information. Net coal production of 1,035 million metric tons, therefore, consists of 143 million metric tons of cleaned coal and 892 million metric tons of raw coal.

The principal emission generated during the coal-cleaning process is solid waste, as indicated in Table 8. The nature of this waste, which is a mixture of ash, coal, and water makes it a serious problem. Because it has a high water content, it is unstable and leads to additional mine acid drainage-type liquids. High coal content leads to smoldering waste pile

TABLE 8 Total Coal Preparation Emissions, 1991 (in millions of metric tons)

Waste Stream	Coal Combustion	Coal Preparation	Total
CH_4	0.03	0.00	0.03
CO_2	10.06	0.00	10.06
SO_2	0.11	0.00	0.11
NO_2	0.05	0.00	0.05
Particulate/ fly ash	0.22	0.10	0.31
Solid waste	1.22	52.84	54.06
Liquid waste	2.73	58.74	61.47

fires, although a well-operated coal cleaning plant should not reject enough coal in the waste solids to support combustion.

2.2.3 Transport

Coal transport is a significant aspect of China's coal industry. Coal is produced mainly in the north and northeast, while coal is used primarily in industrial coastal cities and some inland industrial regions. Most coal is transported by rail and truck, with some water traffic. In 1991, an estimated 558 million metric tons of coal was transported by rail and 399 million metric tons by truck/water (Sinton et al. 1992). The coal transport sector uses primarily liquid fuels (diesel and gasoline) for trucks and water traffic and coal for steam locomotives. Although the number of electric trains in China has increased, they account for less than 5% of transport volume.

The principal emissions in coal transport, therefore, are from diesel fuel and gasoline consumed by trains and trucks. Coal transport emissions are shown in Table 9. The emissions for coal-fired trains are not included in this sector; rather, they are included as a separate category in the end-use combustion sector.

2.2.4 End Use: Combustion

Direct coal combustion is the main source of emissions in China. It is essential to be aware of how different coal combustion is in China compared with other countries. In most countries, large electric utility boilers are the principal coal combustors, whereas in China, electric utility boilers account for only 30% of the total coal use. Industrial boilers and residences use 32% and 16% of China's coal, respectively. (These large percentages are due to the lack of the natural gas and oil alternatives in China. This situation is nearly identical to that in the United States before natural gas and oil became readily available in the 1950s.) Table 10 gives coal consumption data for both combustion and noncombustion end uses.

TABLE 9 Total Coal Transport Emissions, 1991 (in millions of metric tons)

Waste Stream	Liquid Fuel Combustion	Coal Transport	Total
CH_4	0.01	0.00	0.01
CO_2	19.00	0.00	19.00
SO_2	0.02	0.00	0.02
NO_2	0.26	0.00	0.26
Particulate/ fly ash	0.01	0.09	0.11
Solid waste	0.00	4.66	4.66
Liquid waste	0.00	0.93	0.93

Because the quantity of coal consumed by each sector is a key determinant of coal emissions, the distribution published by China's Ministry of Energy was used in the analysis.

Emissions from the end-use sectors vary significantly, primarily because of differences in the coal combustion process (Table 11). Electric utilities use large pulverized-coal boilers that are efficient and have tall stacks. These plants usually have relatively low NO_2 emissions and have methods to control particulates. They are also the most cost-effective plants for potential SO_2 control, specifically those that burn high-sulfur coals. However, China is not seriously interested in controlling SO_2 because such controls increase capital and operating costs but reduce net power output.

TABLE 10 End-Use Coal Consumption, 1991

Stage	Total (millions of metric tons)	Total[a] (%)
Total Combustion	880	82.4
Industrial boilers	350	32.8
Utility boilers	330	30.9
Residences	180	16.9
Rail	20	1.9
Total Noncombustion	188	17.6
Coking	110	10.3
Other	78	7.3
Total end use	1,068	100.0

[a] Values may not add because of rounding.

Source: Ministry of Energy (1992).

TABLE 11 Direct Coal Combustion End-Use Emissions Level, 1991 (in millions of metric tons)

Waste Stream	Rail	Residential Use	Industrial Boilers	Utility Boilers	Total
CH_4	0.01	0.10	0.18	0.12	0.41
CO_2	32.35	291.16	599.45	596.59	1,519.55
SO_2	0.36	2.85	6.24	5.64	15.09
NO_2	0.18	1.66	2.76	2.17	6.77
Particulate/ fly ash	0.81	7.30	8.56	5.77	22.45
Solid waste	4.60	43.83	77.05	66.38	191.86
Liquid waste	9.30	83.66	162.67	153.38	409.00

Industrial boilers are usually small stoker (grate-type) boilers that are less efficient and have short stacks. They usually do not have methods to control particulates or sulfur. Stoker boilers normally specify a special "stoker grade" coal, which is low in fines. However, because the coal mining and pricing system generates low-quality coal, most stoker boilers burn coal that is very high in fines and ash. This practice causes excessive soot emissions of fly ash and unconverted carbon. However, China is currently reforming its state-controlled mining and allocated pricing systems. These reforms should eventually reduce this problem.

China has done innovative work in developing briquettes for residential coal use in cooking. Briquettes greatly reduce emissions of particulates and also capture sulfur when limestone is used in the briquette manufacturing process. However, emissions from residential coal combustion are still a significant problem in the winter. Traditional grate (stoker-type) furnaces with very short stacks are used for heating. Emission factors for the various end-use sectors are compared in Table 12.

2.2.5 End Use: Noncombustion

The principal noncombustion end uses for metallurgical coal are (1) in coke ovens to make coal gas (town gas) and (2) in blast furnaces for iron- and steelmaking. Other uses include cement kilns and coal gasification, which are used to produce town gas and ammonia. These uses indirectly determine emissions of sulfur and particulates.

Although this stage of the coal fuel cycle produces relatively small quantities of emissions (Table 13), they are a serious environmental problem. It is well known that air emissions, liquid waste, and solid waste from coke ovens contain tar, benzene, and phenolic derivatives that are carcinogens. Many of the worst "Superfund" sites in the United States are old coke oven and town gas sites. Modern coal gasification and direct-reduced iron processes totally avoid these environmental problems.

TABLE 12 Direct Coal Combustion End-Use Emission Factors (lb/10^6 Btu)

Waste Stream	Rail	Residential Use	Industrial Boilers	Utility Boilers
CH_4	0.06	0.06	0.06	0.04
CO_2	174.01	174.01	184.25	194.49
SO_2	1.94	1.70	1.92	1.84
NO_2	0.99	0.99	0.85	0.71
Particulate/ fly ash	4.37	4.37	2.63	1.88
Solid waste	24.74	24.74	23.68	21.64
Liquid waste	50.00	50.00	50.00	50.00

TABLE 13 Noncombustion Coal End-Use Emissions, 1991 (in millions of metric tons)

Waste Stream	Coking	Other	Total
CH_4	0.77	0.43	1.21
CO_2	238.67	133.42	372.09
SO_2	0.31	0.14	0.45
NO_2	0.58	0.61	1.19
Particulate/ fly ash	0.92	1.91	2.82
Solid waste	17.47	17.15	34.62
Liquid waste	33.00	7.79	40.79

3 ENVIRONMENTAL IMPACTS FROM CONTINUED RELIANCE ON COAL

The discussion in Section 2 illustrates that coal has played (and continues to play) a critical role in China's economic development. Coal forms the backbone of China's energy structure. It accounts for 73% of the primary commercial energy produced and almost 80% of the primary commercial energy consumed (Fridley 1991; Hulme et al. 1992; Xi and Dowlatabadi 1993). In 1990, coal production totaled 1,080 million metric tons; 44% was produced in state-run mines and 56% in local mines (Fridley 1991; Liu et al. 1992).

China is the world's largest consumer of coal, and almost all sectors use this fuel. However, unlike most industrialized nations, most of China's coal is used for direct burning; only 30% is converted to secondary energy. Industry consumes the largest share of coal (i.e., 36-50%). The power generation and residential sectors also use significant amounts of coal (i.e., 23-26% and 18-22%, respectively). The coking industry and the transportation sector consume smaller quantities of coal (i.e., 7-9% and 2%, respectively) (Fridley 1991; Zhao and Zhang 1991; Liu et al. 1992). Although the percentage of coal consumed by the transportation sector is small, certain modes of transport depend heavily on this fuel, most notably the railway sector. Coal provides up to 90% of the fuel used by railways in China (Sathaye and Goldman 1991).

In China, the electricity-generating sector has grown faster than all other users of energy since the 1980s. Much of this increase has been fueled by coal, both through expanding coal-generated electric capacity and by replacing oil-fired plants with coal. In 1988, coal provided fuel for 87-95% of all thermal power generation, which, in turn, accounted for 80% of the total power generated (Liu et al. 1992). Most thermal power generating plants in China are small — only one-third have a thermal generating capacity of 200 MW or larger (Liu et al. 1992). Most of these plants are located near either industrial centers or coal resources in the east, northeast, and north.

The distribution of energy production and consumption in China is geographically disproportionate, which means that energy resources (mostly coal) must be transported long distances. Most of the coal is initially transported by rail: in 1985, more than 60% of all coal mined in China was transported by rail (Haugland and Roland 1990). Conversely, coal is a significant commodity on all transport modes; it accounts for 41-50% of the nation's rail capacity, 21% of trucking traffic, and 20% of barge cargo (Liu et al. 1992; Wilson 1993). Coal is transported primarily from mining regions of northern, central, and northeastern China to populated, industrialized regions to the east and south (Doyle 1987). This spatial separation is illustrated by the fact that freight charges account for almost 80% of the total delivered price of coal (Wilson 1993). Lack of coal transport capacity is largely responsible for current energy shortages in China. In addition to being the major rail cargo, coal also accounts for 81% of the energy used by the railway system (Liu et al. 1992).

Given China's past and current dependence on coal, which has resulted in an infrastructure built around coal use, its large coal reserves, and its limited reserves of other energy sources, coal will continue to dominate the country's economy. Even with China's high consumption rates, its vast coal reserves should last several centuries (Travis 1991). Demand for coal is likely to continue to increase dramatically as economic liberalization and deregulation stimulate economic development in the next several decades.

China's heavy dependence on coal has come at a high price. Coal is widely regarded as a "dirty" fuel. The extraction, preparation, and combustion activities associated with its use generate pollutants and lead to adverse environmental effects. Coal combustion alone accounts for up to 80% of the air pollution in China (Travis 1991). In addition, the coal fuel cycle contributes significantly to China's water and land pollution, as well as a significant portion of radioactive releases, water consumption, and land-use changes.

Although the quality of Chinese coal varies significantly, its physical nature and general characteristics are responsible for the nation's environmental problems. In general, Chinese coal has a high ash content (an average of 20%, with a range of 3-50%) and low-to-medium sulfur concentrations (0.5-3%) (Liu et al. 1992). Some coals, particularly from southern regions, have sulfur contents as high as 7%, but these regions produce only minimal amounts of the coal mined in China. Significant quantities of coking coals are used as boiler fuel in China, whereas these coals are commonly used in steelmaking in other countries. Use of coking coal in boilers releases greater amounts of air, water, and solid pollutants because of the higher proportion of volatile organic compounds (VOCs) and ash released compared with that of other coal types (Liu et al. 1992). The relatively low heat value of Chinese coals also results in greater emissions of air pollutants and other contaminants per unit of energy compared with other countries.

3.1 ENVIRONMENTAL IMPACTS ASSOCIATED WITH THE COAL FUEL CYCLE IN CHINA

Each step in the coal fuel cycle is associated with various adverse impacts on the environment and human health. The most common adverse impacts are the effects on the atmosphere, water systems, and soil and land resources.

In general, the most serious disruptions to the environment from the coal fuel cycle result from extraction and consumption activities, but negative impacts can also result from processing, transport, transmission, and postconsumption activities. In China, the most serious consequences of coal use probably result from the way in which coal is consumed. The environmental impacts associated with each step of this cycle are illustrated in Figures 5-12.

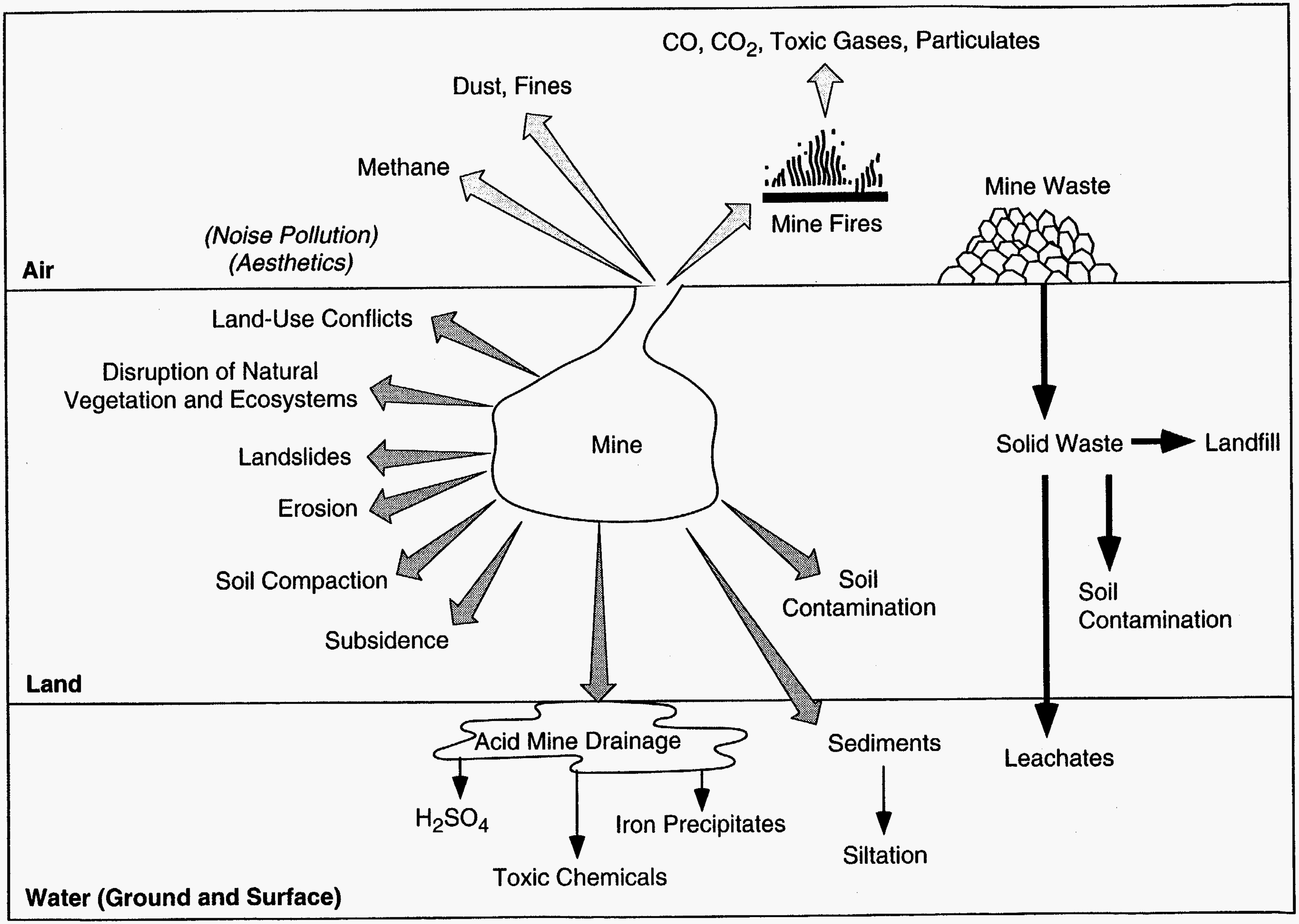

FIGURE 5 Step 1 in the Coal Fuel Cycle: Extraction

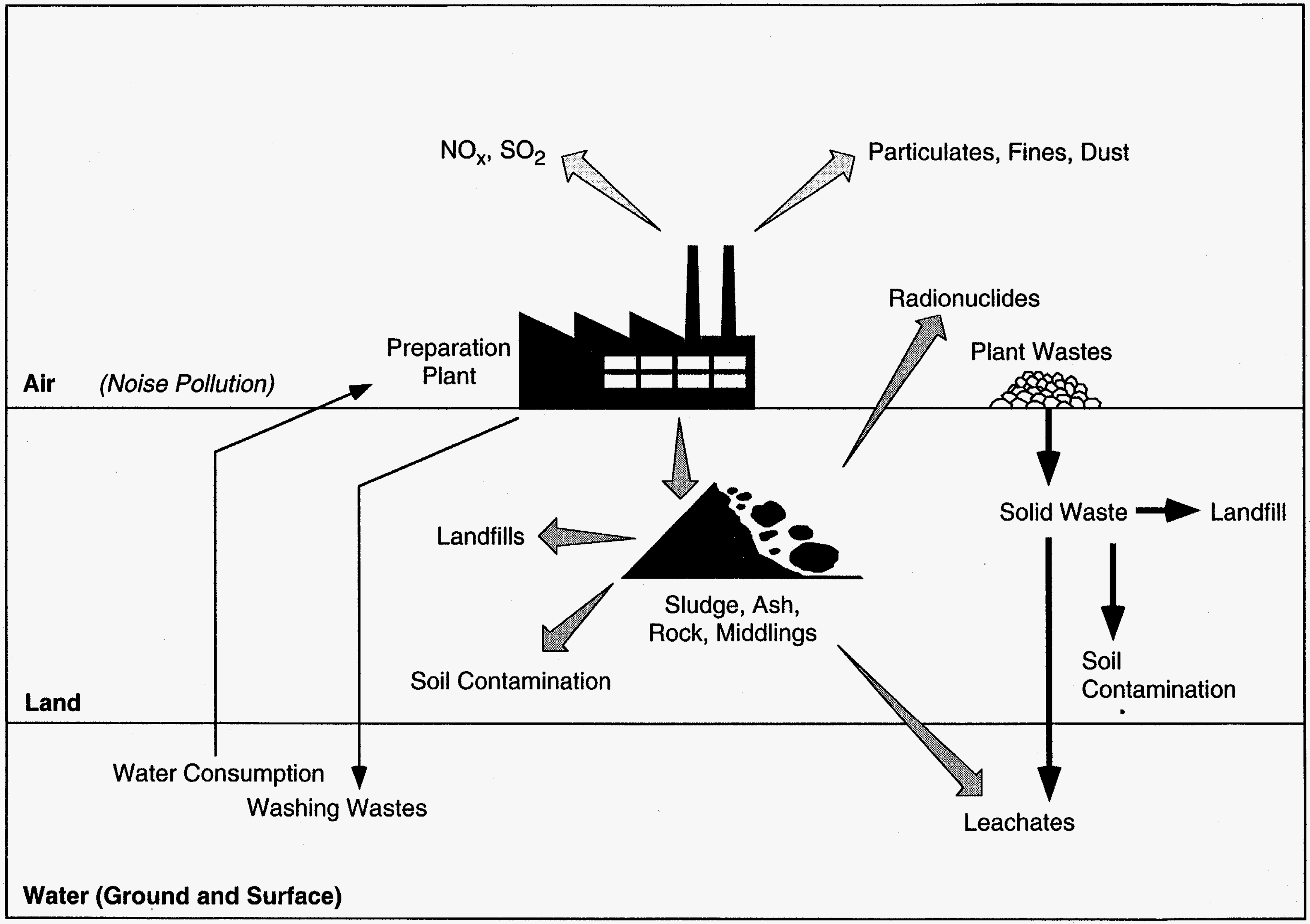

FIGURE 6 Step 2 in the Coal Fuel Cycle: Preparation and Beneficiation

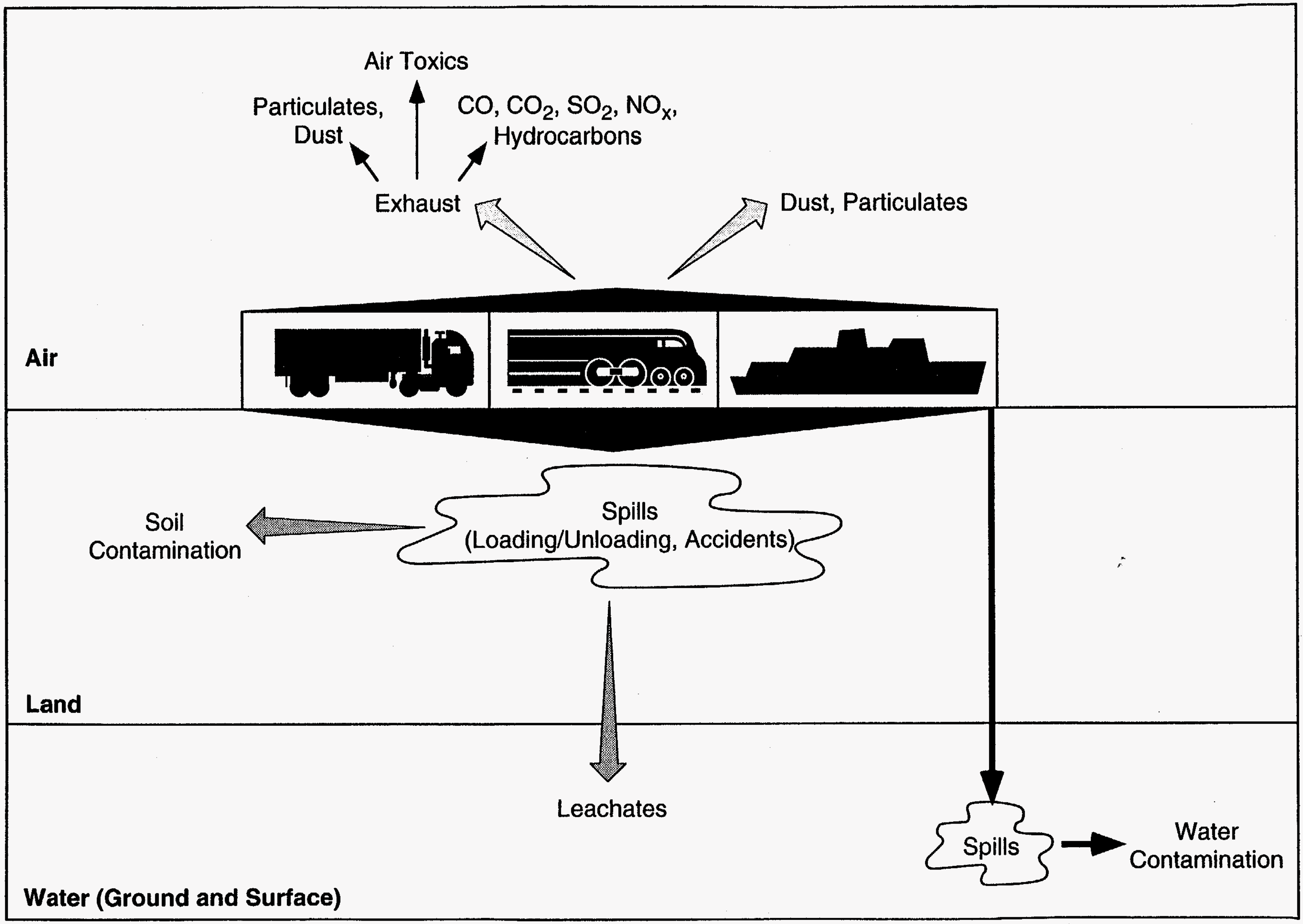

FIGURE 7 Step 3 in the Coal Fuel Cycle: Transport

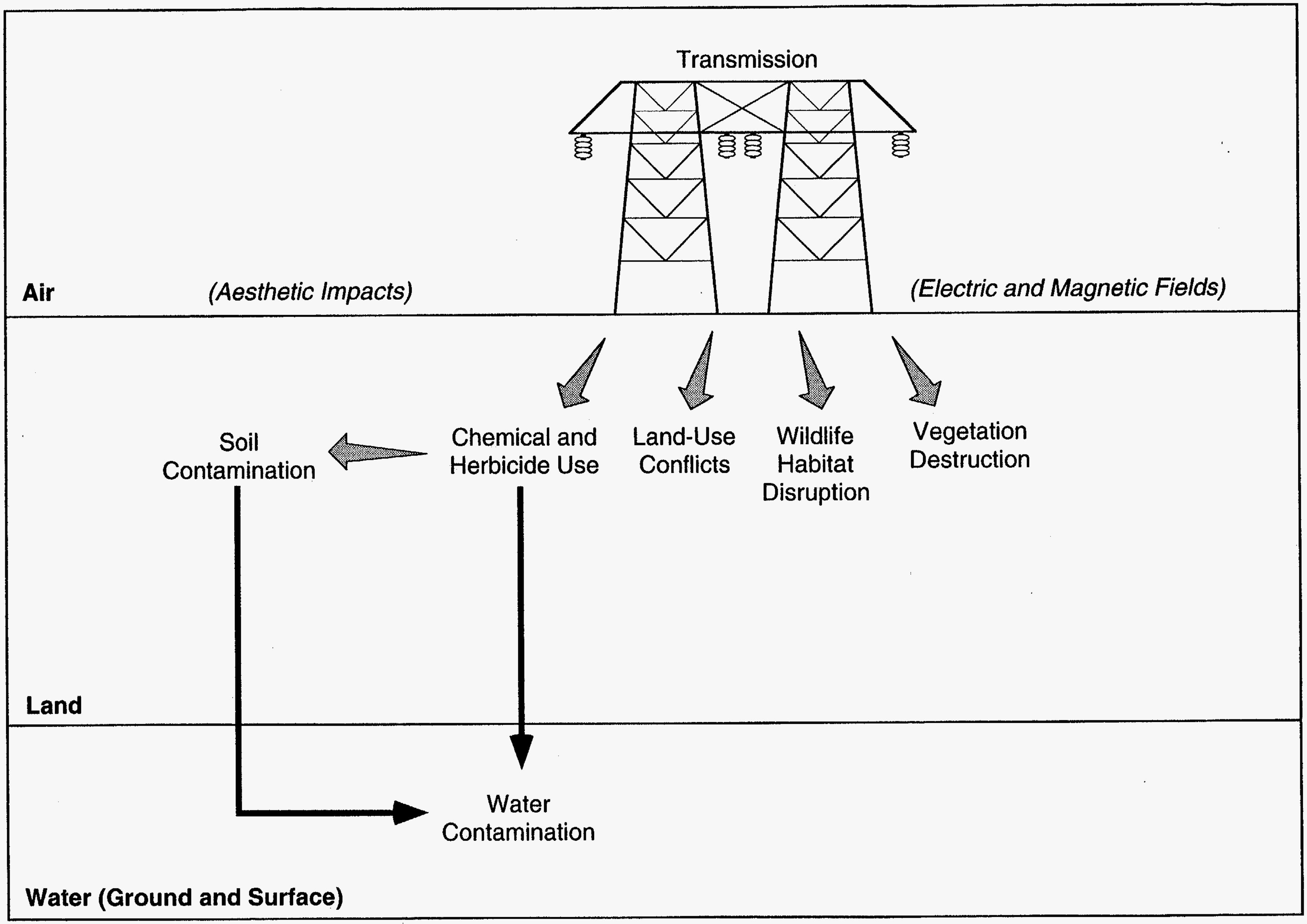

FIGURE 8 Step 4 in the Coal Fuel Cycle: Transmission

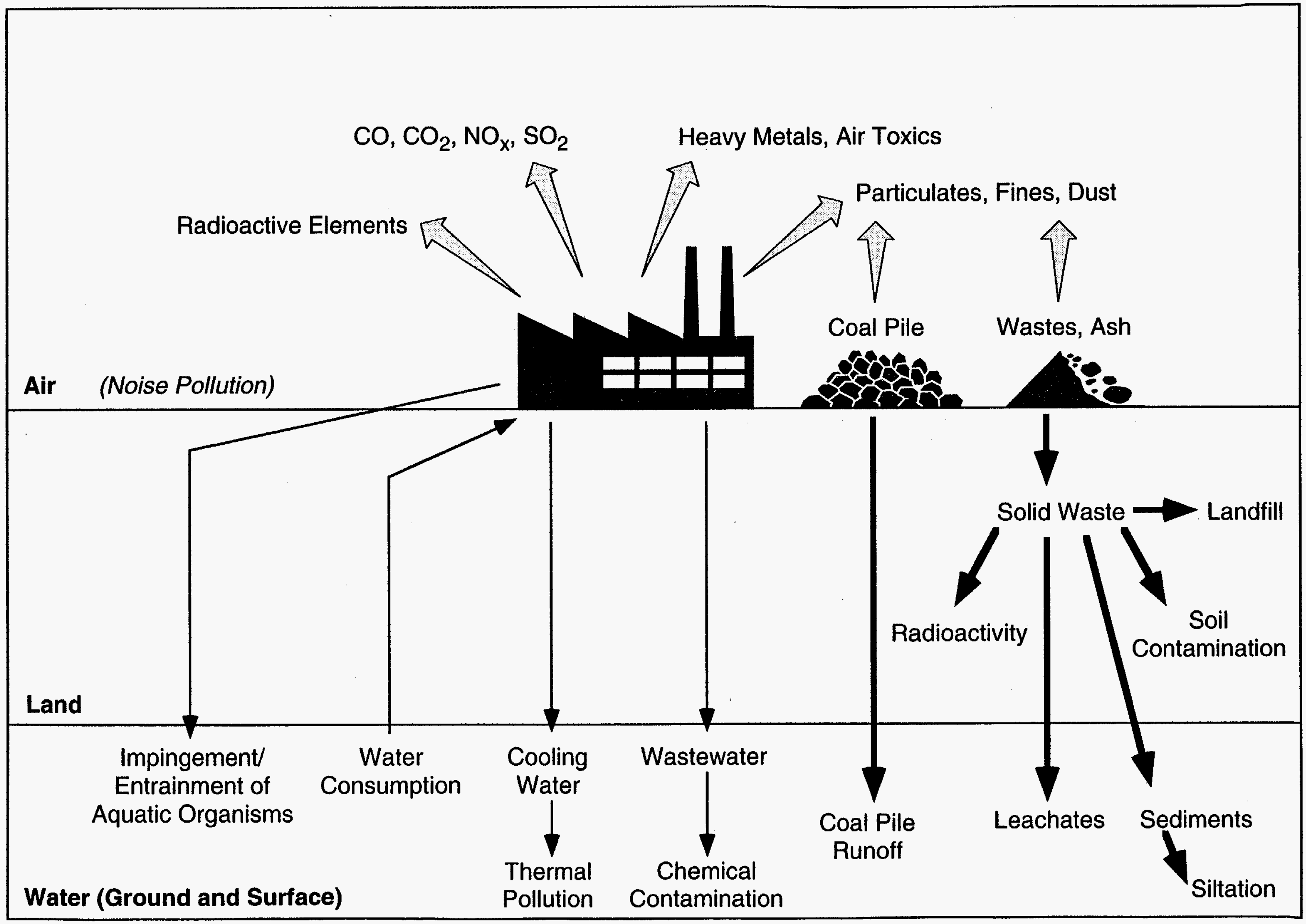

FIGURE 9 Step 5a in the Coal Fuel Cycle: Consumption — Utility and Industrial Boilers

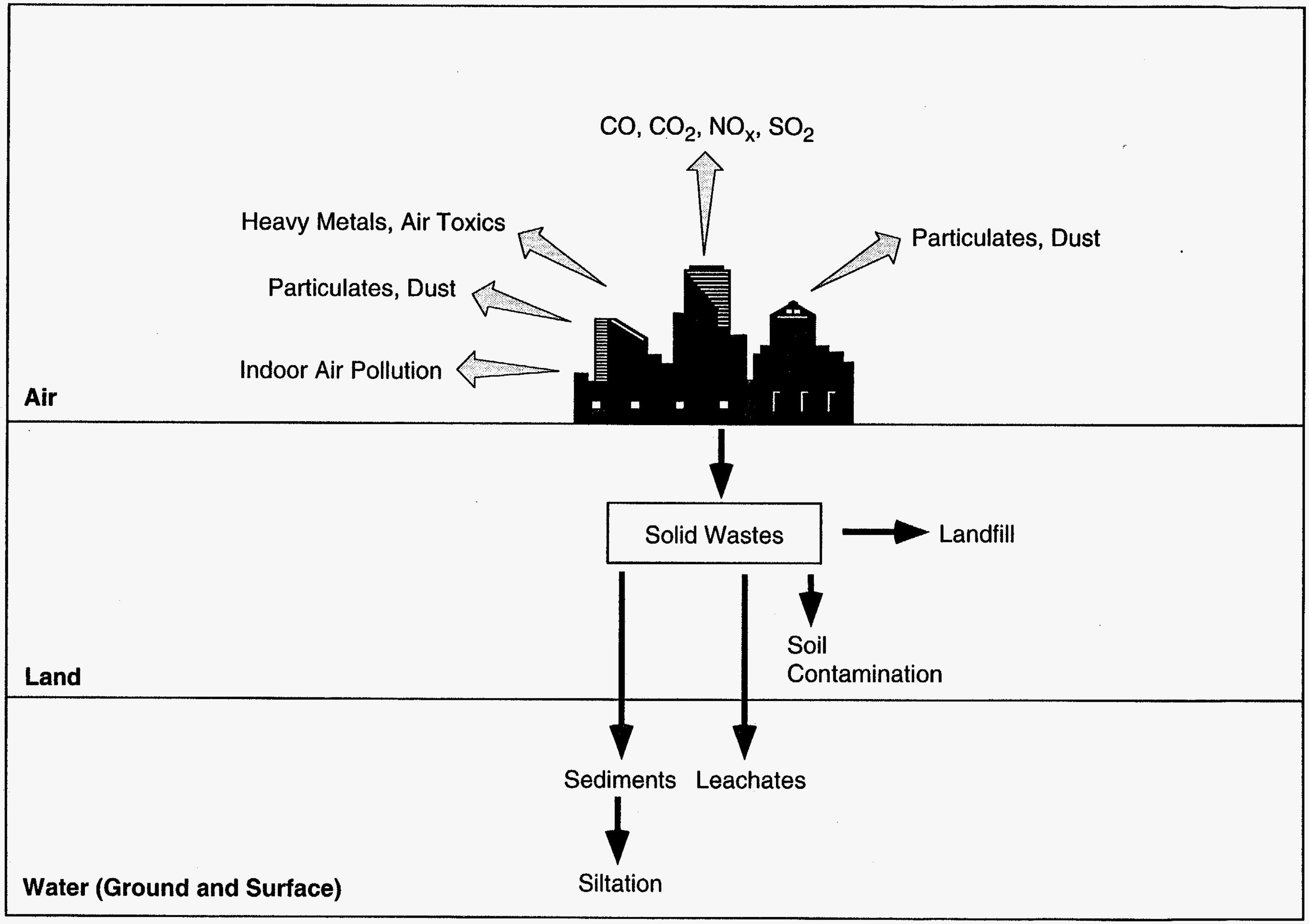

FIGURE 10 Step 5b in the Coal Fuel Cycle: Consumption — Commercial and Residential Boilers

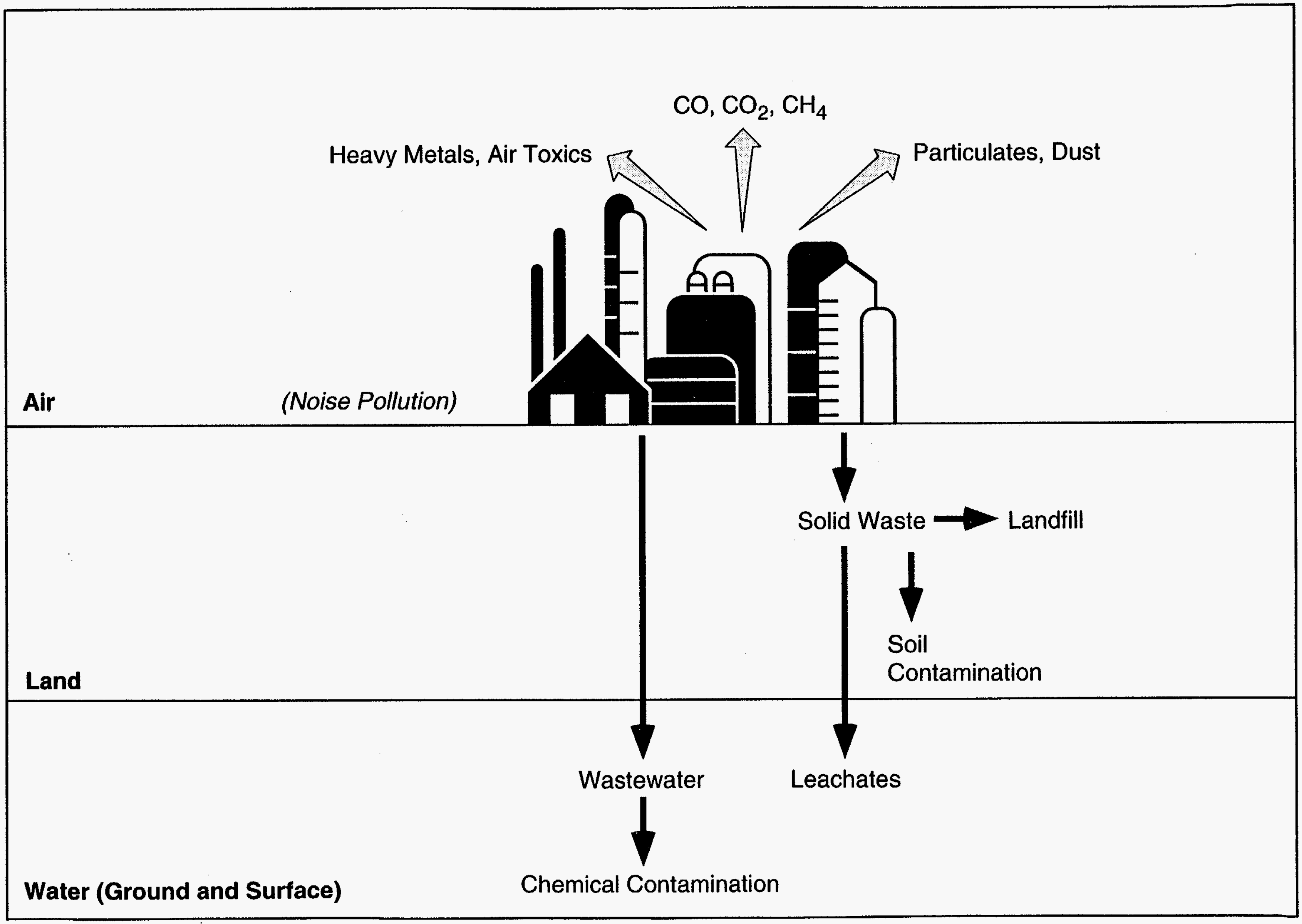

FIGURE 11 Step 5c in the Coal Fuel Cycle: Consumption — Industrial: Coke Ovens

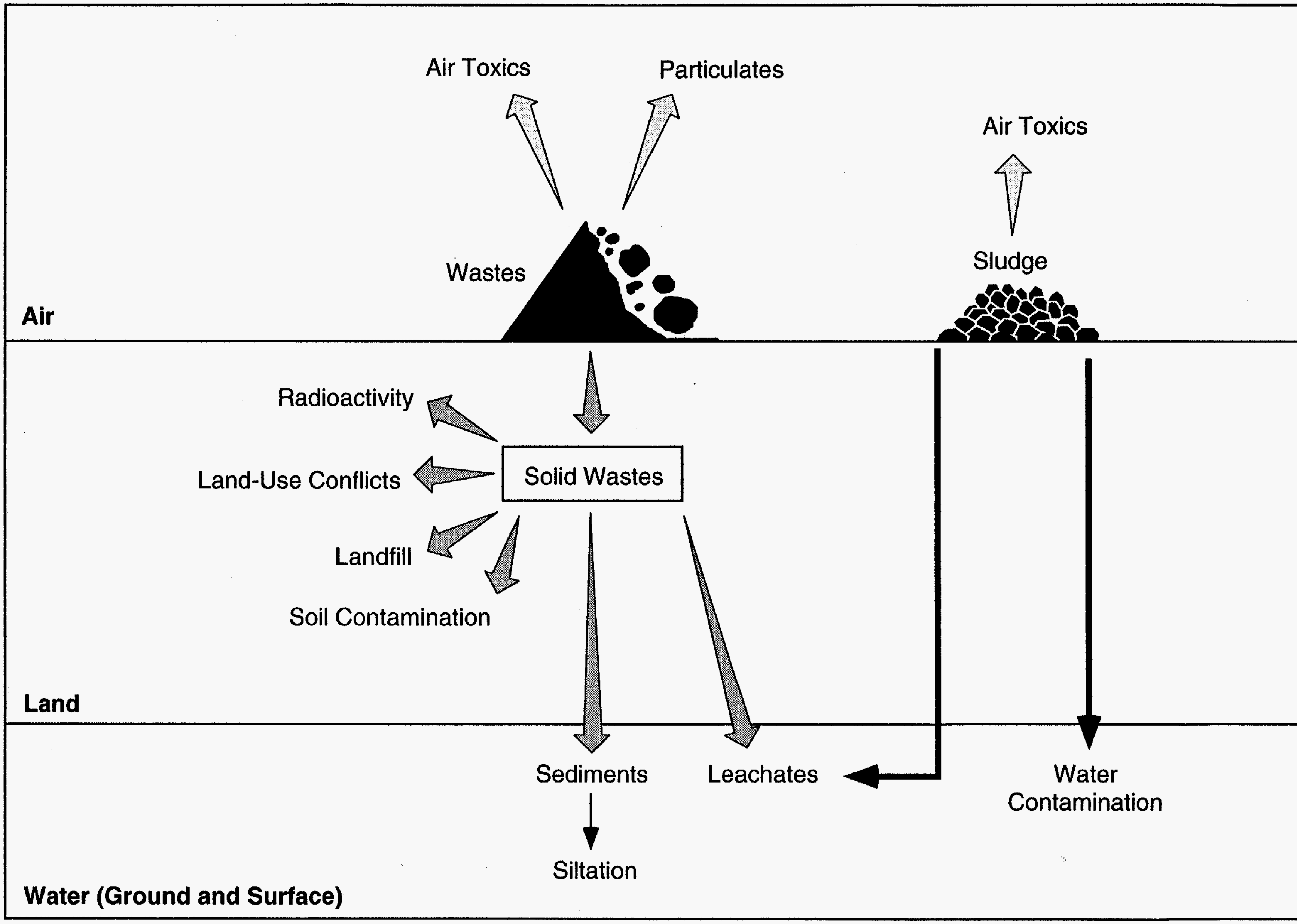

FIGURE 12 Step 6 in the Coal Fuel Cycle: Postconsumption

3.2 GENERAL IMPACT OF THE COAL FUEL CYCLE ON VARIOUS MEDIA

The steps in the coal fuel cycle can affect all major components of the environment, including the atmosphere, groundwater and surface water, and soil and terrestrial resources. These impacts are discussed in detail in Sections 3.2.1-3.2.4. These sections concentrate on the ways in which coal is extracted, prepared, transported, and consumed.

3.2.1 Air Pollution

The majority of coal consumed in China is burned unwashed, unsorted, and unscrubbed. Only 17-18% of raw coal produced in 1989 was washed to any extent (Sinton 1992). In addition, postcombustion pollution control devices are extremely rare in China. Only some power-generating facilities, specifically those with a capacity of more than 200 MW and those located in or near large cities, are equipped with electrostatic precipitators, which reduce emissions of particulates. Emissions from large power plants are also usually released from tall stacks, thus dispersing and diluting air pollutants over large areas. The paucity of coal beneficiation, the high ash content of Chinese coals, and the general lack of postcombustion controls in most sectors result in the release of large quantities of pollutants into the air.

Much air pollution, particularly in urban areas, results from unwashed and unsorted coal consumed in small, inefficient residential burners and industrial boilers with no emission controls. In regions with the highest need for winter heating, most residential buildings are heated with small coal-burning boilers and stoves (Liu et al. 1992). In addition to being very inefficient (efficiency values of between 10 and 18%), such devices emit significant quantities of pollutants and have very low release heights. Coal provides 80% of the commercial fuel used in residential households in China (U.S. Senate 1993). Approximately 53% of coal (and 90% of the coal used as a heating source) is consumed in urban areas (Aling 1993; Lu 1993). This consumption pattern produces large amounts of ground-level air pollutants in populated areas. Such concentrations significantly reduce the air quality in urban areas and often result in life-threatening indoor air quality.

Coal combustion has contributed 75% of the total dust (14-17 $\times 10^6$ ton/yr) emitted from anthropogenic sources into the atmosphere in China (Fengqi 1992). The average concentration of total suspended particulates (TSPs) in urban areas is 432 $\mu g/m^3$. Some northern cities can average more than 2,700 $\mu g/m^3$. (The U.S. primary ambient air quality standard for particulate matter [annual mean] is 75 $\mu g/m^3$.) Coal combustion is also responsible for 90% of the estimated 15-19 $\times 10^6$ tons of SO_2 released from anthropogenic activities (not including biomass-related emissions) into the atmosphere (Institute of Energy Economics, Japan 1989; Xi et al. 1990; Travis 1991). In the south and southwest, where coal has a relatively high sulfur content, and in northern cities where large quantities of coal are burned in space heaters in the winter, ambient levels of SO_2 can be very high. In southern cities, average SO_2 concentrations vary between 20 and 450 $\mu g/m^3$, and in northern cities, such concentrations can average up to 380 $\mu g/m^3$ and have been measured at more than

770 μg/m^3. (The U.S. primary ambient air quality standard [annual mean] for SO_2 is 80 μg/m^3.) Coal combustion also accounts for approximately two-thirds of the anthropogenic NO_x released into the atmosphere in China (the total amount of NO_x emitted in 1982 was approximately 4.1×10^6 tons).

Both SO_2 and NO_x emissions can lead to the formation of acidic deposition; this phenomenon has been observed in south and southwestern China, where the pH of precipitation averages 5.0, and individual precipitation events with pH as low as 2.25 have been measured (Harte 1983).

Coal combustion also releases VOCs and hydrocarbons, especially from inefficient combustion processes, which are common in China. Numerous trace elements are also present in coal, and many are released into the atmosphere during combustion. Some of these elements, especially heavy metals, are toxic if emitted in sufficient quantities. Given the enormous quantities of coal consumed in China, particularly in urban areas, it is possible that these elements will reach hazardous levels in some parts of the country. Little information is available on ambient levels of these substances because, to date, toxic pollutants have not been measured routinely.

The significant quantities of NO_x and VOCs emitted in China, along with the weather patterns prevalent in much of the country, are very conducive to the formation of ground-level ozone. Although ozone concentrations have not been systematically monitored to date, it is likely that ozone levels are fairly high in many parts of China, particularly in urban areas. For example, time-series data gathered in the 1980s for the city of Lanzhou indicated that ozone levels averaged 170 μg/m^3 in the winter months, with peaks as high as 210 μg/m^3 (Siddiqi and Zhang 1984; Tian and Zhou 1991). (The U.S. maximum hourly standard for ozone is 120 μg/m^3.)

Coal also contains various radionuclides that can be released upon combustion. Again, because of the large quantities of coal consumed in China, the resulting emissions could threaten human populations and the natural environment in some parts of the country. It has been estimated that many coal power plants routinely release more radiation than the average nuclear power facility (Hall et al. 1986).

Chinese coals also have a very high fluorine content (2,000-3,000 parts per million [ppm]). Coals from the eastern United States typically have a fluorine content of 80 ppm. Combustion of high-fluorine coals, especially in stoves and space heaters, can result in high ambient levels of this substance, especially in indoor environments (Travis 1991).

Finally, combustion of coal generates significant quantities of CO_2; coal combustion releases more CO_2 per unit of energy produced than any other commercial fuel. Given the vast amount of coal used in China, it is not surprising that approximately 85% of China's commercial-energy-related CO_2 emissions (which account for 11-12% of the world's total CO_2 emissions from commercial energy sources and cement manufacturing) result from coal use (Sathaye and Goldman 1991; Hulme et al. 1992; U.S. Senate 1993).

Because most coal in China is produced far from the places of consumption, large amounts of coal must be transported long distances. Much coal is transported by rail, with the remainder by barge and truck. This high transportation activity, coupled with the lack of pollution control equipment on transport vehicles, releases considerable air pollutants.

Because the coal fuel cycle releases large amounts of various pollutants, particularly in urban areas, many of these areas routinely exceed the World Meteorological Organization's ambient air quality standards by five to six times (WuDunn 1993). For example, concentrations of particulates in Shanxi province are often 20 times the maximum U.S. urban allowable limit (U.S. Senate 1993), and SO_2 and particulate levels in many Chinese cities are among the highest in the world. The air in many of China's northern cities, particularly in winter, resembles that found in London and the Meuse Valley in Belgium during the severe "smog" incidents of the 1940s and 1950s.

In addition, because the nation relies heavily on coal for cooking and space heating in residential and commercial buildings, indoor air quality is a serious problem and significantly threatens human health. Indoor air quality has not been studied in depth to date, but preliminary studies estimate that indoor TSP concentrations typically average 450 $\mu g/m^3$, with some studies indicating indoor levels as high as 2,000 $\mu g/m^3$ (Florig 1993). Because much of the population spends most of its time indoors (i.e., on average 90%), the poor quality of indoor air poses serious consequences for human health.

Large quantities of CH_4 are also released in China (specifically from underground coal mines). In addition to posing an immediate occupational danger, CH_4 contributes to the buildup of greenhouse gases. Also, fires frequently occur in abandoned and, to a lesser extent, in functioning mines. These fires result from coal wastes left behind or combustible gases that build up in sealed-off mines. While these fires threaten the surrounding human population, they also release CH_4, CO, CO_2, toxic gases, and particulates into the air.

3.2.2 Water Pollution

Coal production and consumption activities may also negatively affect China's hydrological cycle. In a country where water shortages are already common and where uneven distribution, both temporally and spatially, often leads to floods and droughts, water resources contaminated as a result of coal use can have many negative effects.

Almost every step in the coal fuel cycle requires considerable quantities of water. The coal extraction process significantly affects surface and underground water supplies. These impacts include acid mine drainage, siltation of streams and rivers due to increased erosion, and disruption and contamination of underground aquifers. The most significant contamination of surface waters and aquifers associated with coal extraction activities is related to acid mine drainage. This drainage occurs when precipitation (and any other) runoff enters mine shafts and reacts with the sulfur and other compounds in the coal seam. The reaction forms sulfuric acid and iron precipitates (Hall et al. 1986). Thus, sulfuric acid can drain into nearby waterways and adjacent aquifers.

Activities associated with preparing and processing coal can also contaminate water supplies, particularly if coal washing and other beneficiation activities are used. Transmission of power generated by coal-fired power plants indirectly affects water resources, mainly through the use of chemical herbicides for right-of-way routes that can leach into adjacent water systems.

Consumption of coal, especially by utilities and industries, often generates large quantities of wastewater. Some of this water is contaminated with residues produced in the coal combustion process. In addition, cooling water used by industrial and power plants is usually treated chemically to control corrosion, scale formation, plant growth, and pH. Electric utilities also generate wastewater in the de-ashing process. Boiler blowdown water adds to pollutant loadings of water systems. The wastewater released from power plants often contains high levels of toxic metals, acids, bases, oils, and other harmful chemicals. Although recently built power plants are sometimes equipped with water treatment plants, most older and smaller power generating units and industrial plants usually discharge untreated wastewater (Zhai 1992). It is estimated that 72% of all wastewater from utility and industrial facilities in China is discharged into rivers, lakes, and the ocean without treatment (Fengqi 1992).

Further contamination of groundwater and surface water occurs from coal piles stored at power-generating plants and coal mining and preparation sites. Precipitation percolating through these coal storage and waste piles can leach toxic substances from the piles and carry contaminants into surrounding surface and groundwater systems.

In addition to chemically polluting water, coal-fired power plants also pollute water resources by releasing cooling water used by thermal power plants. This cooling water is substantially warmer than when it was drawn from water systems. This increase in water temperature can disrupt fish population dynamics, change species composition, reduce wetland plant communities, increase concentrations of algae, and reduce biodiversity. Damage to aquatic organisms can also occur because of impingement and entrainment of these organisms as cooling water is pumped into power plant intake pipes.

3.2.3 Land Impacts

The activities associated with producing and consuming coal also negatively affect land. Whereas air and water are most affected by chemically induced alterations associated with the coal fuel cycle, land is physically altered. The coal fuel cycle produces significant quantities of contaminated solid wastes. These wastes are generated during mining and preparation operations as well as during actual consumption of the coal.

Coal mining wastes include mine spoils and tailings, mineral debris, and mine overburden. Solid wastes produced as a result of coal preparation activities include coal ash, mineral debris, and dust and fines. Consumption of coal generates fly ash, bottom ash, boiler slag, and solid cleaning wastes. Solid wastes produced in the coal fuel cycle are composed primarily of silicon, aluminum, iron, and calcium but can also contain small concentrations of toxic substances such as arsenic, barium, cadmium, chromium, lead, mercury, and selenium. Because China's coal has a high ash content and is not cleaned before combustion, coal combustion generates significant amounts of solid wastes. In addition, solid wastes generated while using coal can contain radioactive substances; for example, much radioactivity is present in fly ash. The proportion of toxic and radioactive substances produced in the coal fuel cycle is small compared with the total quantity of solid wastes generated. However, China uses large quantities of coal, which produce high absolute amounts of solid wastes. Solid wastes are also created if postcombustion pollution control devices are installed on coal combustion plants. However, except for electrostatic precipitators, such control devices are still rare in China.

Solid wastes generated in the coal fuel cycle must be disposed of. In China, many of these wastes are disposed of in landfills or deposited in surface impoundments. Such sites do not provide high-level protection from leaching of toxic substances from the waste sites into surrounding soils, surface water, and groundwater. In addition to contaminating water and soils, activities associated with the extraction, preparation, and consumption of coal can negatively affect terrestrial systems in other ways. Significant potential exists for subsidence of land as a result of the extensive underground coal mining activities common in coal-rich regions of China. Such subsidence can lead to loss of human life and damage to man-made structures and natural ecosystems. This damage is especially significant in heavily populated areas. Thus, it is of considerable importance in China, given its high population density. It has been estimated that 1×10^9 m^2 of land is currently undergoing subsidence because of coal mining (Fengqi 1992).

Other activities associated with the coal fuel cycle also require significant amounts of land (e.g., the mines, coal storage areas, power plant sites, transmission lines, landfills, wastewater treatment ponds, evaporation ponds, and coal preparation sites). These land requirements can result in land-use conflicts, particularly in heavily populated and agricultural regions, where land is already a precious commodity.

3.2.4 Other Adverse Environmental Effects

Other adverse impacts associated with the coal fuel cycle in China include the following:

- Soil erosion and compaction from coal mining and storage activities;
- Noise pollution from the operation of machinery;
- Occupational hazards related to the extraction, transportation, preparation, consumption, and waste disposal activities associated with coal use;
- Localized weather changes due to thermal releases by coal-fired power and industrial plants;

- Electromagnetic radiation exposure from transmission lines; and
- Ecological displacements and changes from increased activities associated with coal mining and power plant/industrial activities.

3.3 NATIONAL AND REGIONAL IMPACTS OF POLLUTANTS IN CHINA

Society is usually not concerned about emissions and discharges of pollutants until these pollutants come into contact with and/or alter human populations and valuable resources. Thus, it is necessary to consider not only the magnitude of pollutant emissions and discharges, but also the location of such releases relative to human settlements and valuable resources. In addition, the vulnerability of such resources and populations to the pollutants of concern must be considered. China exhibits significant differences in the spatial distribution of the various activities associated with the coal fuel cycle and the resources and populations potentially affected by these activities.

3.3.1 Spatial Distribution of Various Steps in the Coal Fuel Cycle

Most of the mining and extraction activities associated with coal production in China take place in the north central regions, particularly in the provinces of Shanxi, Shandong, Hebei, and Henan, and in the northern province of Heilongjiang. Figure 13 shows the location of the major coal producing mines in China. In most cases, little of the coal is processed and consumed at the mine. Most of the coal produced in China is transported to other regions.

Specifically, coal is transported to the east and south, where the bulk of the coal is consumed. Coal used by industry and power plants generally occurs in a crescent-shaped region that extends from northeast of Beijing along the central coast to Shanghai. A few industrial and utility plants are also located in southwestern China near Chongking in the province of Sichuan. Figures 14 and 15 depict the locations of the majority of China's industrial and utility plants, respectively. Residential and commercial consumption of coal is centered in the urban areas, which largely coincides with the crescent-shaped area described above.

Given this spatial distribution of production and consumption of coal within China, it is evident that major coal transportation routes are from north central regions to Beijing and the major urban areas along the central coast. The railway and barge maps shown in Figures 16 and 17, respectively, illustrate the coal transport routes. Coal flows are depicted in Figure 4.

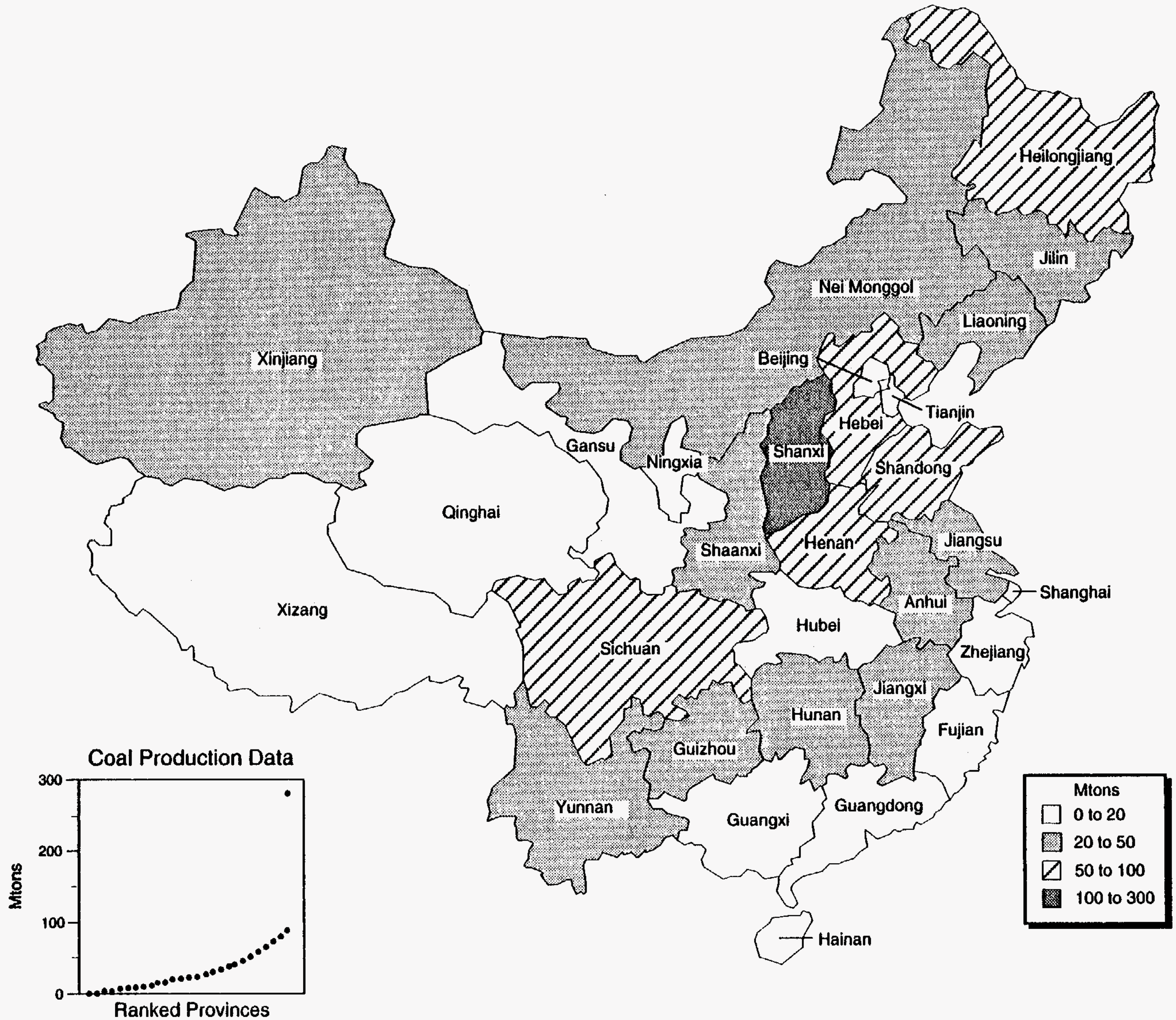

FIGURE 13 Major Coal-Producing Provinces in China, 1989 (Source: Adapted from Sinton 1992)

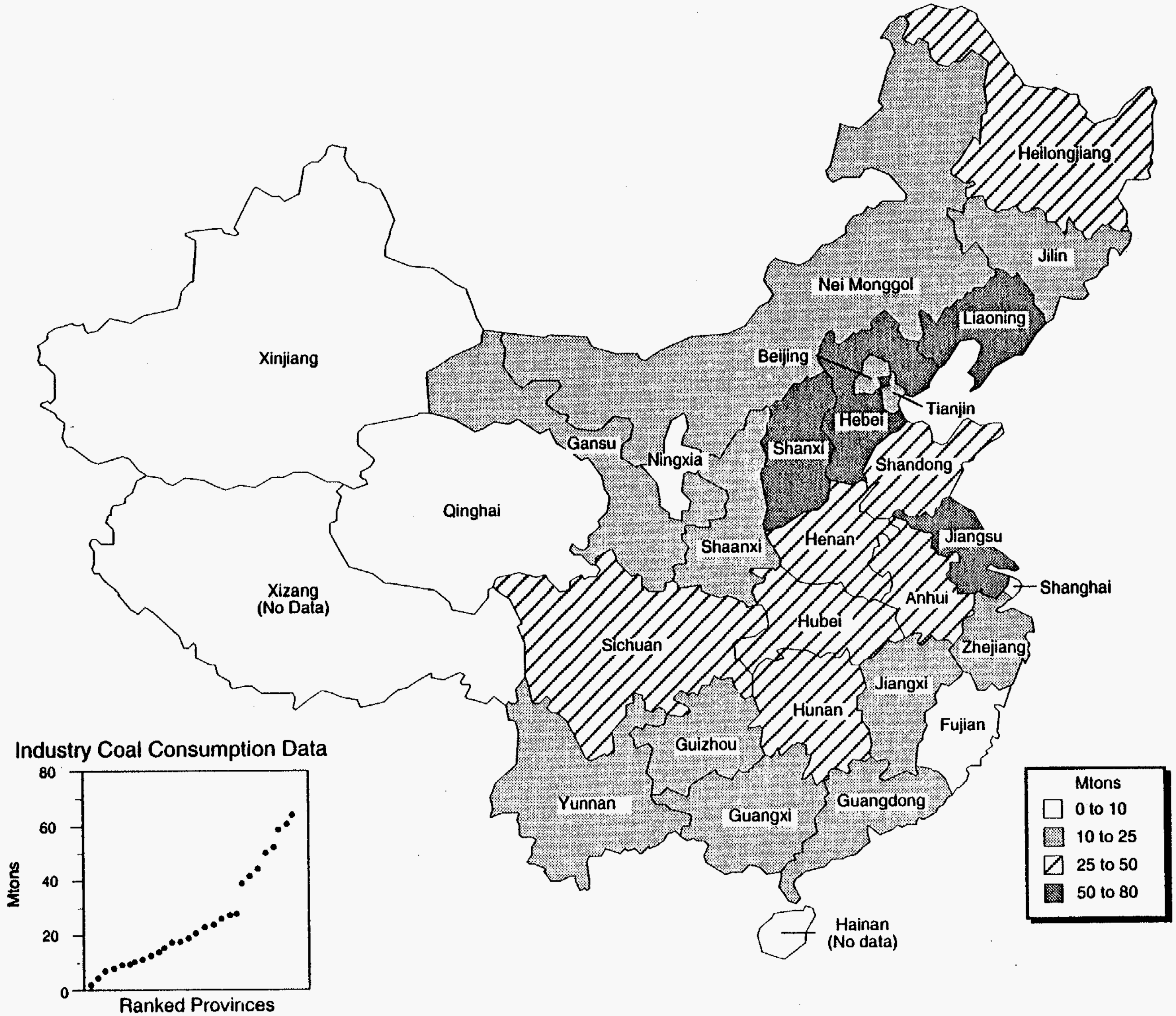

FIGURE 14 Location of China's Major Industrial Coal Consumers, 1988 (Source: Adapted from Sinton 1992)

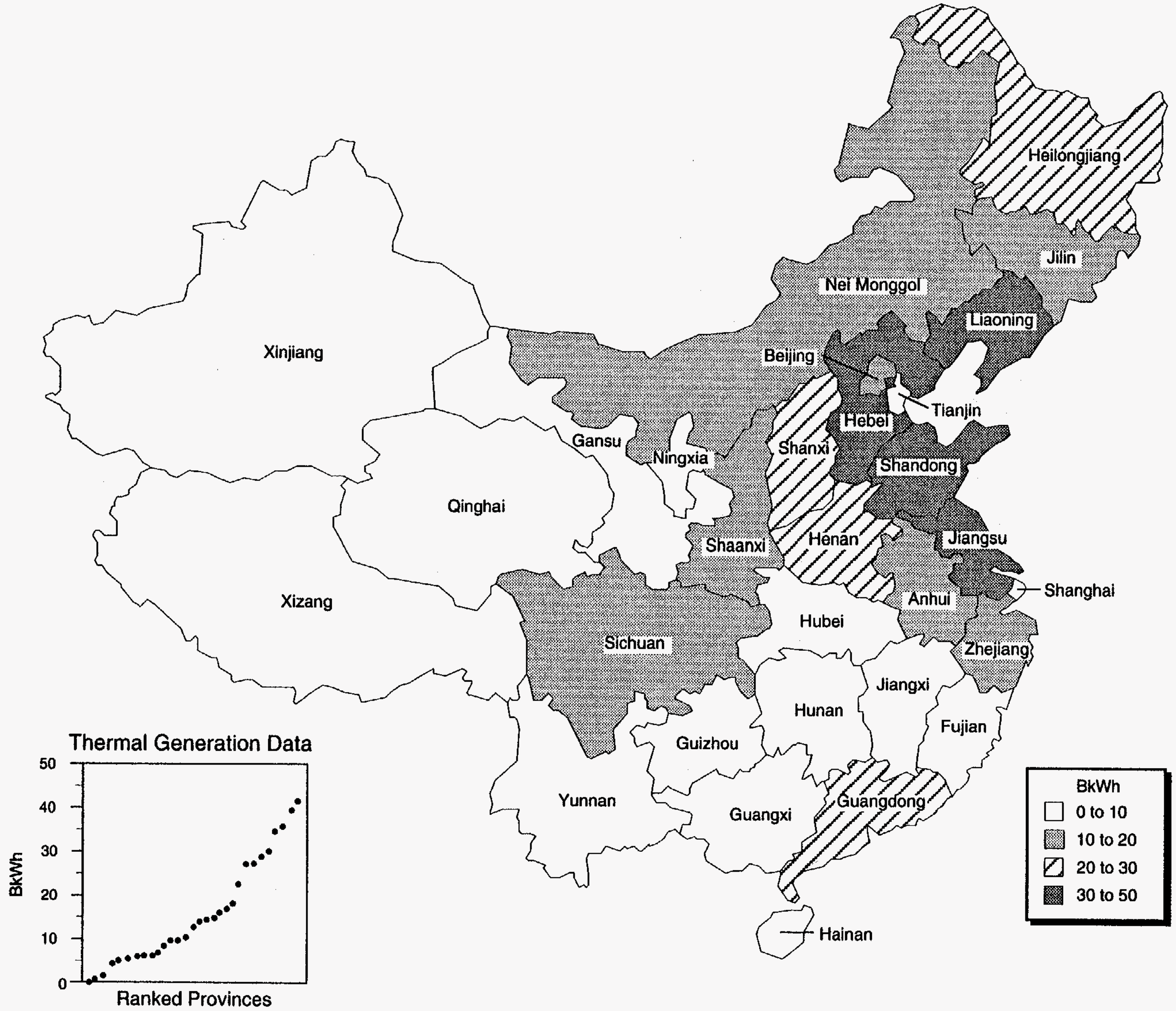

FIGURE 15 Location of China's Thermal Electricity Generation Plants, 1989 (Source: Adapted from Sinton 1992)

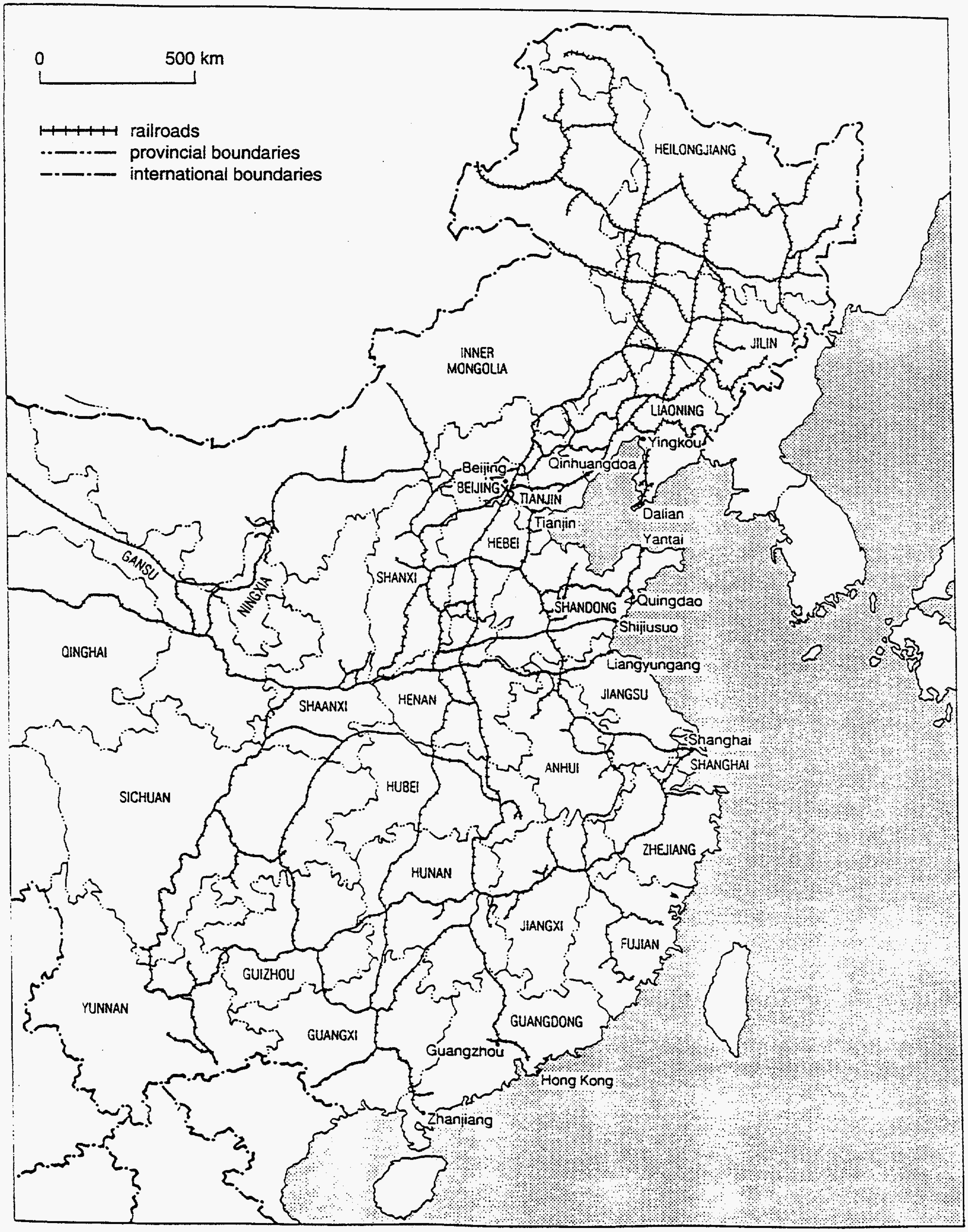

FIGURE 16 China's Coal Transport Infrastructure: Rail Network (Source: Adapted from Doyle 1987)

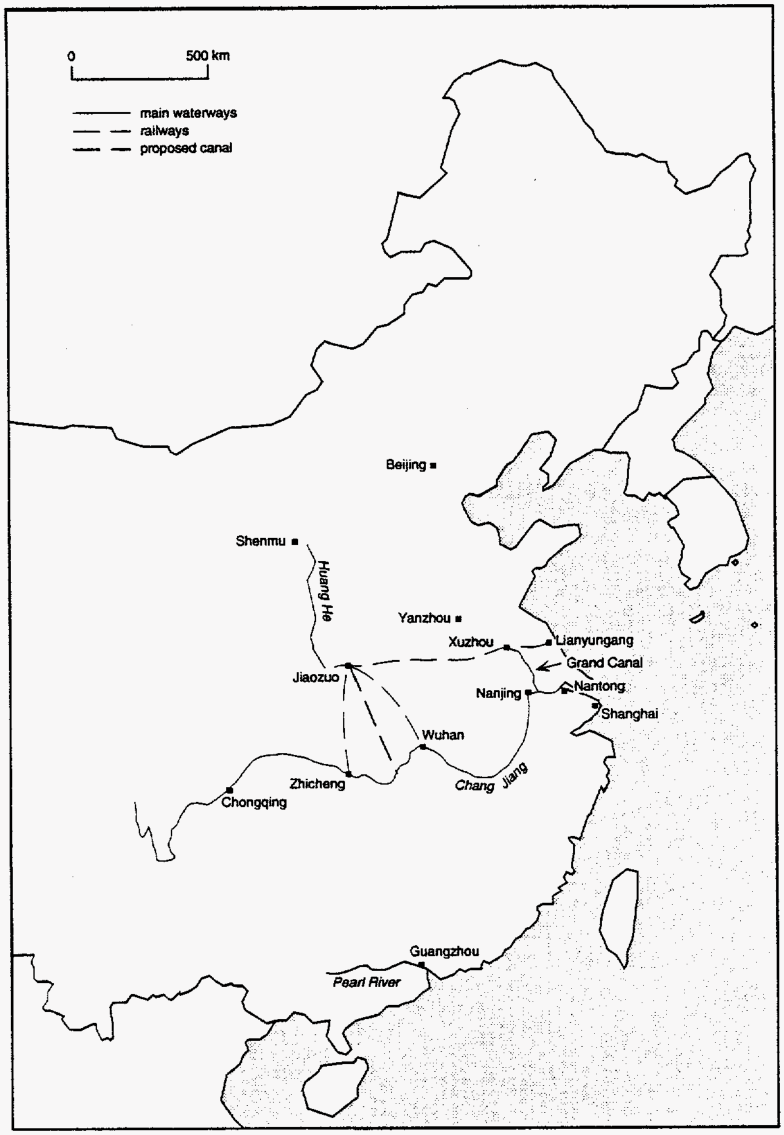

FIGURE 17 China's Coal Transport Infrastructure: Main Coal-Carrying Waterways (Source: Adapted from Doyle 1987)

3.3.2 Specific Impacts Associated with the Coal Fuel Cycle and Their Spatial Distribution

The supply and consumption pattern associated with the coal fuel cycle results in a separation of emissions and discharges and their related impacts. This separation occurs because the various activities of the coal fuel cycle generate different kinds of pollutants, and thus affect human populations and natural resources in different ways.

Most water- and land-related impacts associated with the coal fuel cycle result from mining/extraction and postconsumption activities; only minor contributions result from power-plant and industrial combustion processes. Because of the spatial distribution of these activities, most of the water and land pollution problems associated with coal use are likely to occur in the north central mining regions and along the east central coast. The degradation of water quality and accumulation of mine, sludge, and postcombustion wastes are likely to present the most serious consequences in areas already experiencing water shortages, water pollution problems, and land-use conflicts. These areas include land surrounding the major urban centers of eastern China, the arid regions in the vicinity of mines in the Shanxi and north central China, and the farming communities of central China.

3.3.2.1 Health Impacts

It is likely that the greatest risks to human health from the externalities associated with coal consumption are due to emissions of particulates and SO_2, ozone formation, and degradation of indoor air quality. Other risks are due to (1) the presence of toxic substances in drinking water and foodstuffs from surface water and groundwater and (2) soil contamination from extraction and postconsumption activities. Moreover, climate changes due to increased concentrations of greenhouse gases could also negatively affect human health. For example, higher temperatures could exacerbate air quality in many regions, and changes in climate could result in "new" diseases and disease vectors to which indigenous populations have little resistance.

One of the common air pollutants with a well-established link to increased mortality and morbidity is TSP matter. Studies conducted in the United States have indicated that TSPs increase chronic background mortality by approximately 0.5% to more than 6% for each 100 $\mu g/m^3$ increase in ambient TSPs (Florig 1993).

The leading cause of death in China is respiratory disease; respiratory illnesses accounted for approximately 26% of deaths in 1988. The incidence of lung cancer follows a distinct north-south gradient, which correlates to the prevalence of using coal for heating and cooking (Florig 1993). In addition, lung cancer rates in many regions of China are higher in women than they are in men, probably because women generally spend more time indoors than men. It is likely that air pollution (indoor and outdoor) is a significant factor in these mortality patterns.

Morbidity patterns in China also implicate deterioration in air quality, both outdoors and indoors. Chronic obstructive lung diseases are common in China and seem to increase as concentrations of TSPs increase. Some researchers have concluded that 50-60% of all cases of upper respiratory dysfunction in urban areas in China result from particulates (Xu et al. 1991). Sulfur dioxide and NO_x also contribute to respiratory illness, particularly in susceptible individuals (e.g., the young, the elderly, asthmatics).

Water quality is also poor in most of China, especially in industrial regions. Many of these same areas also experience water shortages, which poses risks to human health. Most rivers in urban areas are highly polluted because untreated sewage is usually discharged directly into waterways. The World Bank has estimated that only one in seven persons has access to safe drinking water in China (Wilson 1993). In addition, harvests of shellfish from coastal regions have declined dramatically during the past decade, and any seafood likely contains high levels of toxic substances.

In general, the greatest risks to human health from activities related to the coal fuel cycle are likely to occur in the highly populated region in the crescent-shaped area highlighted in Figure 18. This is due to the concentration of large numbers of susceptible people in this region, coupled with the prevalence of combustion activities (leading to high concentrations of air pollutants) and shortages and pollution of water supplies in this area.

3.3.2.2 Agricultural Impacts

The agricultural resources most at risk from coal fuel cycle activities are those located in the north central mining regions as well as those grown on farmlands adjacent to

FIGURE 18 Main Coal Consumption Regions and Population Centers in China

urban areas in northern and coastal regions. Agricultural crops are threatened primarily by the following:

- Particulates generated by coal mining and combustion activities,
- High ambient concentrations of SO_2 and tropospheric ozone from coal burning in urban areas,
- Acidic precipitation formed from long-range transport of SO_2 and NO_x released from coal consumption activities, and
- Water and soil contamination from extraction and postcombustion processes.

These problems are likely to exacerbate the water shortages and land-use conflicts that currently threaten agricultural production in these parts of China. In addition to adverse impacts on croplands in regions close to coal fuel cycle activities, it is also possible that because air pollutants are transported over great distances, agricultural resources downwind (and downstream) from such activities could also be at risk. Wilson (1993) estimated that damage to agricultural crops, forest resources, and buildings in China from acidic deposition alone totaled $2.8 billion in 1991.

During the summer, the monsoons carry pollutants from urban areas in eastern China into the western provinces (Bhatti et al. 1992). This transport pattern applies particularly to the precursors of acidic deposition (SO_2 and NO_x) and ozone. Rainfall with pH as low as 2.25 has been observed in the remote Tibetan plateau of Qinghai province (Harte 1983). Because rice and wheat are susceptible to low pH rain (pH < 4.5) (Zhao and Xiong 1988), coal-related activities that release large amounts of acidic deposition precursors (especially SO_2 and NO_x) could affect major agricultural resources in areas far from where such activities take place.

In addition, SO_2 and ozone individually or in combination with each other and other pollutants can adversely affect crop growth and yield. It is well known that both of these pollutants directly and indirectly damage plant species — in fact, ozone pollution is the strongest link between air pollution and plant damage, particularly with respect to sensitive species such as tobacco, soybeans, corn, and citrus fruits. In some cases, SO_2 and ozone have greater-than-additive (synergistic) effects on crops (e.g., soybeans). This effect probably occurs because ozone exposure can damage the cell membrane, thereby increasing the permeability and leakiness of these membranes to various molecules, including SO_2. Conversely, SO_2 can lead to wider than normal opening of the stomata of many plant species, which, in turn, can make it easier for ozone to enter the cell. Synergistic interactions between SO_2 and NO_x have also been observed; SO_2 inhibits the ability of plants to detoxify nitrate formed in plant cells. Because the highest concentrations of these pollutants occur in urban areas where many of the combustion activities that generate such pollutants take place, agricultural resources near these urban centers (Figure 18) are at highest risk from such impacts.

Some pollutants, especially nitrogen compounds, can fertilize agricultural crops. Nitrogen is a basic crop nutrient and is often limiting in many locations.

3.3.2.3 Forest Resources

China's forest resources occur primarily in the northeast and west. At present, given the spatial distribution of the coal fuel cycle activities, only the forests in the northeast are likely to be threatened to any significant degree by the pollutants and discharges that result from coal use. Most of this threat occurs because air pollutants are transported from the urban areas in northeastern China. However, as is the case for agricultural resources, it is possible that summer monsoons could carry pollutants from eastern China into the forested regions of the western mountains and plateaus (Bhatti et al. 1992). As mentioned, acidic deposition observed in the Tibetan plateau in western China (which is far removed from any major coal producing or consuming regions) could threaten the forest resources in this region.

Ozone has also been shown to be damaging to many tree species, especially pines, which are a major component of many forest ecosystems in China. Ozone pollution is not confined to the source of its precursor pollutants; rather, like acidic deposition, ozone is a regional pollutant, and it affects locations far removed from its origin. Given the direction of the prevailing winds in China, forests in northeastern China will probably be most at risk from the effects of tropospheric ozone pollution. As is the case for agricultural crops, interactive effects of different pollutants and of various pollutants with pest and disease organisms can also adversely affect forest resources. Because trees are long-lived plants, cumulative effects of pollutants such as SO_2, NO_x, and ozone also have important impacts on forest resources. In addition, some air pollutants, especially nitrogen, can have fertilizing effects on forest resources.

3.3.2.4 Climate Change Impacts

Climate changes are also directly related to the coal fuel cycle. Extracting, processing, and using coal are likely to be the major anthropogenic sources of greenhouse gases in China and a significant contributor to global greenhouse gases. The impacts associated with climate changes could potentially affect resources throughout China. However, these impacts are likely to be felt more seriously in marginal areas, such as those currently experiencing severe water deficiencies, resources at the extreme limits of their natural ranges, populations and resources in coastal and low-lying areas and those located on permafrost, and populations in areas already stressed by overcrowding and pollution.

Changes in temperature and precipitation would probably significantly affect the distribution of vegetation within China. Regions currently at the extremes of their natural range would be most seriously affected. These regions include the temperate coniferous forest of the northeast, the temperate steppe of central China, and the warm temperate deciduous forests of the central coastal region. Thus, climate changes could alter many currently forested areas by converting these areas to shrubland or grassland.

The agricultural sector would probably experience the most significant impacts from climatic change. For example, increased temperatures could allow many regions to change from single to double and from double to triple cropping systems, thereby increasing agricultural production. However, such potential increases imply the availability of an adequate water supply. Even if rainfall were to increase in some parts of the country, it is unlikely that an adequate water supply would be available to sustain an increase in agricultural production. Furthermore, warmer temperatures would increase evapotranspiration and dry up current water reservoirs. WuDunn (1992) and Wilson (1993) have estimated that a rise of 1-2°C would reduce agricultural production in China by 5%. In addition, sea-level rise would increase land-use pressures, leaving less land available for agriculture and result in saltwater intrusion of aquifers, thereby reducing the amount of water available for irrigation. Changed climatic conditions would alter the distribution of pests and disease vectors, which could seriously affect monoculture crops that have little resistance to these exotic diseases or pests.

In addition to the changes in the distribution of natural ecosystems and agricultural crops, alteration of weather patterns and sea-level rise could considerably affect China's water supply; coastal areas, which are the backbone of the Chinese economy; air quality; the transportation infrastructure; buildings, especially those built on areas currently covered by permafrost; and human health. In general, the resources at highest risk from climatic changes are those located in (1) the crescent-shaped region shown in Figure 18, (2) other coastal areas, and (3) the Tibetan plateau.

4 OVERVIEW AND ASSESSMENT OF THE TRANSFERABILITY OF AVAILABLE EXTERNALITY COST ESTIMATES TO CHINA

4.1 OVERVIEW OF EXTERNALITIES

Section 3 examined the types of adverse impacts associated with the coal fuel cycle. The type and extent of such impacts are functions of the affected environmental medium and the activities in question (e.g., fuel extraction, residential consumption, industrial uses). In situations in which adverse impacts are not factored into the decision-making process, such impacts take on special significance. The term *externalities* is used to refer to "the case where an action of one economic agent affects the utility or production possibilities of another in a way that is not reflected in the market place" (Just et al. 1982, p. 269). As this definition suggests, externalities can be either positive or negative. Adverse effects of pollution are a common example of a negative externality. From an economic perspective, considering the effects of externalities is motivated by the fact that, in the absence of market intervention, externalities lead to inefficient market outcomes.

As discussed in Section 3, coal use in China adversely affects air, water, and land resources. These impacts generally result from pollutants, including various types of air emissions (e.g., SO_2, NO_x, particulates, CO_2, and so forth). In addition to polluting activities, uses of land and water resources (e.g., siting production facilities and disposing of solid wastes) and the use of water for cooling also adversely affect these environmental media. These adverse impacts impose direct and indirect costs on society. Potential sources of direct costs include adverse health effects, loss of recreational opportunities, impaired visibility, and reduced agricultural yields. Indirect costs include possible employment and income effects that result from direct costs.

The treatment of external costs usually focuses on the effects of pollution from production processes. However, as shown in Section 3, household activities (e.g., using coal for heating) can also result in external costs. Coal use by various sectors in an economy — commercial, industrial, and public (government) — has the potential to impose external costs on society. In addition, external costs can be incurred at different points in the fuel cycle. Failure to include external costs in the decision-making process results in an excessive level of output. For example, the proportion of coal-fired generation of electricity may be excessive when compared with the amount of electricity produced. Furthermore, the amount of coal used to heat homes and support industrial production activities may exceed the economically efficient amounts. In each case, resources are overallocated to the production of the consumer good in question.

For simplicity, the following discussion focuses on the use of coal in generating electricity. However, the basic principles could be applied to any uses of coal noted here and in Section 3.

4.1.1 Economic Theory of Externalities

According to the simple market model (i.e., the model of supply and demand), the equilibrium quantity of a good is that quantity at which consumers' marginal willingness to pay equals the minimum price demanded by producers. The demand schedule for a good represents the marginal benefits of each additional unit of output, and the supply curve represents the marginal costs of production. Hence, the market equilibrium condition can be restated as follows: produce the level of output at which marginal benefits equal marginal costs. This condition is the underlying principle of microeconomic analysis.

When all the costs and benefits of production and consumption of a good are reflected in the respective supply-and-demand curves, the market equilibrium is socially efficient. In turn, socially efficient resources will be allocated to the production of the good. However, if production or consumption results in third-party costs, the market-determined outcome will be inefficient.

Consider the following situation based on an electric utility. Manufacturing a good produces pollution (e.g., air emissions). In making production decisions, the managers of the utility consider internal costs of production such as payments for labor and raw materials. Such costs are commonly referred to as "marginal private costs." The utility finds it profitable to produce additional units of output as long as the willingness to pay for the marginal unit of output exceeds its marginal private costs. The pollution generated as a by-product of production and released into the atmosphere also results in costs in the form of damages to the environment or human health and welfare. However, in the absence of outside pressure (e.g., regulation or legal sanctions), the utility has no incentive to consider these costs in its decision-making process. These costs are therefore external to the utility. However, they are borne by society.

The sum of marginal private costs and marginal external costs is referred to as "marginal social cost." This sum is the correct measure of costs to use in determining the socially efficient level of output. Because under the privately determined equilibrium, marginal private costs equal marginal benefits, marginal social costs exceed marginal benefits by the amount of the marginal external costs. Therefore, the appropriate response is to decrease output sufficiently so that costs decline until marginal social costs and marginal benefits are equal.

These concepts are presented in Figure 19, which illustrates the supply-and-demand conditions for good X, the production of which results in pollution. The supply curve S_p reflects only the marginal private costs of production. The supply curve S_s reflects the sum of marginal private costs and marginal external costs. Hence, the vertical distance between the two curves represents marginal external costs. According to Figure 19, marginal external costs are assumed to increase with the level of output. The privately determined equilibrium price and level of output are P_p and Q_p, respectively. When all marginal external costs are accounted for, the equilibrium output level is Q_s. Thus, Figure 19 confirms an earlier observation: failure to consider external costs in the decision-making process results in an excessive equilibrium output level. Too many resources are devoted to the production of X.

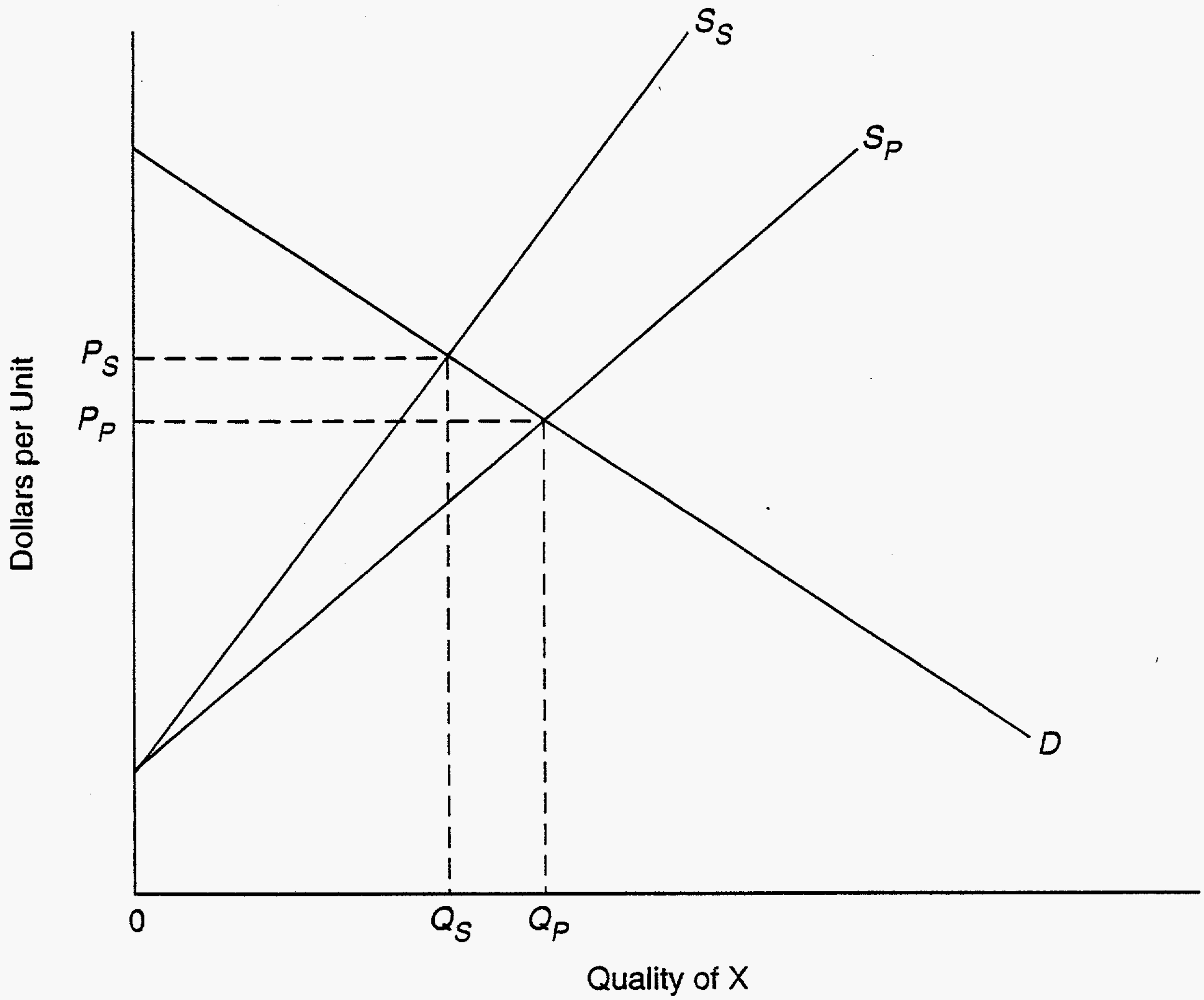

FIGURE 19 Supply-and-Demand Conditions for Good *X*

Actions that force firms to account for some of the external costs of production, such as direct regulation, affect both marginal private and marginal external costs. For example, the mandated use of scrubbers in the electric utility industry would increase marginal private costs because of the costs of operation and maintenance associated with the scrubbers. At the same time, because scrubbers reduce emissions released into the atmosphere, marginal external costs would be reduced, if all things are equal.

4.1.2 Policy Options for Controlling Externalities

Because of the nature of external costs, decision makers (e.g., managers of a firm or the head of a household) have no incentive to consider such costs in the decision-making process. Thus, reducing the externality requires some form of market intervention. In the United States and many other countries, the most frequent response has been to pass legislation that mandates the development of specific regulations designed to reduce the amount and type of pollutants emitted to the environment or to otherwise mitigate activities that result in external costs. In addition to direct regulation, policymakers can also use market-based approaches, including the use of charges and permits. The use of charges requires affected entities to pay a charge (tax) for each unit of emissions that they release into the environment. Under a permit system, permits are issued to affected firms. These permits allow the holder to emit a predetermined amount of a specific pollutant. The number of permits is determined by the regulatory authority and, in effect, limits the total amount of the pollutant that can be emitted during a specific period. Permits can be bought and sold.

From an economic perspective, the principle difference between direct regulation and incentive-based systems is that the latter cost effectively allocates pollution control, while, in most cases, the former does not. Because direct regulation treats firms as equals, no consideration is given to possible differences in control costs across affected firms. Thus, potential economies in pollution control cannot be exploited.

Under both a charge and a permit system, pollution control is the responsibility of those affected parties that can control pollution at the least cost. Consider first the use of charges. In deciding how much pollution to emit (or conversely how much to control), each firm compares the marginal costs of pollution control to the charge. As long as the marginal cost of control is less than the charge, it is in the firm's interest to control that unit of pollution. For those units of pollution for which the marginal control costs exceed the charge, the least-cost solution is to emit the pollution and pay the charge. Assuming that each of the affected firms applies this decision rule, marginal control costs will be equal across the affected firms. In this case, control is not reallocated, which could lower the total cost of controls.

A system of charges yields essentially the same result. Firms control additional units of pollution as long as the marginal costs are less than the market price of permits. Permits are only held for those units of pollution for which marginal control costs exceed the price of the permit.

In addition to the efficiency of direct regulation versus market-based incentives, these approaches share another important difference. Specifically, direct regulation does not internalize all external costs, whereas an incentive-based approach does. Moreover, direct regulation could require all affected firms to reduce pollution by the same percentage or install a particular technology. Direct regulation only forces firms to consider the costs of units of pollution that are actually controlled. Any pollution that continues to be emitted is not factored into the decision-making process. Thus, the marginal costs of production do not reflect any external costs attributable to this remaining pollution.

In contrast to direct regulation, a charge or permit system provides an explicit price for each unit of pollution that the firm continues to emit. Thus, the firm must pay a charge or purchase a permit for each unit of emissions. These costs must be considered explicitly in the firm's decision-making process. This fact has important implications for treating external costs in the policymaking process.

4.1.3 Use of Externality Adders in the Electric Utility Industry

In recent years, a growing number of states have taken actions to account for the external costs of production in the electric utility industry. Such actions range from explicitly quantifying the external costs attributable to various sources of electricity (e.g., coal-fired generating units) to qualitatively considering the possible effects of external costs on the "true" marginal cost of electricity and the relative marginal costs of alternative production techniques. The validity of these "externality adders" and the data used to construct them have been questioned. Analysts have also pointed out that, in some cases, a danger exists for double-counting external costs and, in so doing, overstating the true marginal costs of production.

One criticism of particular interest in this report concerns the methods used to estimate certain external costs. The most common methods include the use of mitigation costs (i.e., the costs of offsetting the adverse effects of pollution), control costs (the costs of pollution control), and damage costs. A second issue concerns whether and to what extent adders should be used in specific situations. As the following discussion illustrates, the method used to estimate external costs and the approach used to examine the current regulatory structure have important implications for the validity of those estimates.

4.1.3.1 Mitigation Costs

One approach to estimating the costs attributable to a pollutant is to consider the costs that would be incurred to mitigate or avert the potential damages. For example, potential damages to agricultural crops might be averted by increasing the use of fertilizer, water, or some other input (Cropper and Oates 1992). The value of the additional resources required to offset the potential damage constitutes the damage costs attributable to the pollutant. That is, the additional production costs constitute the damages incurred by farmers. However, in many cases, opportunities to fully offset the adverse effects of a pollutant are unavailable, and hence, this approach cannot be used.

4.1.3.2 Control Costs

Some analysts have suggested that pollution control costs already incurred provide insights to the marginal costs of certain pollutants. This assertion is based on the argument that because policymakers have selected the existing level of control, it is reasonable to assume that the marginal damages avoided are equal to or exceed the marginal costs of control (i.e., policymakers are behaving in an efficiency-enhancing manner). However, this argument is not based on sound logic. In most instances, pollution standards are not based on economic efficiency. For example, according to the Clean Air Act, the standards for criteria pollutants are to be set to ensure the health and safety of the affected population. The act specifically states that costs are not to be considered in setting these standards.

4.1.3.3 Damage Function Approach

Under the damage function approach, the amount of damage and resulting value of that damage attributable to a particular pollutant are determined by the interaction of a series of usually complex steps. Step 1 involves relating the pollutant in question to specific types of damages. Step 2 empirically estimates the relationship between the quantity of the pollutant and the resulting level of damages. Step 3 uses the relationship estimated in step 2 to estimate the level of damages associated with a given level of pollutant. Step 4 assigns a monetary value to the damages calculated in step 3. Each of these steps is likely to be data intensive. In addition, considerable uncertainty is likely regarding the functional form of the damage function and the estimates of the monetary value of the damages.

Data limitations (with respect to both quantity and quality) have the potential to affect each of the four steps. For example, to be able to relate a pollutant to various damages, all of the potential damages that could be linked to the pollutant must be identified. However, in many cases, latency is a problem (i.e, certain adverse effects are not manifested for many years). Thus, a considerable time lag may result between exposure to a pollutant and the occurrence of adverse effects. Limited data also affect estimates of the damage function and, therefore, estimates of the damage attributable to a given quantity (e.g., concentration) of a pollutant. Finally, limited data on willingness to pay to avoid damages adversely affect the estimation of the monetary value of the damages.

4.1.4 Evaluation of Methods

Of the methods described here, the theoretically valid approach to estimate the external costs of pollution is to rely on damage functions and measures of the corresponding willingness to pay for a reduction in damage. (This approach is anthropocentric; that is, it assumes that all values should be based on the relationship between a specific damage and its effects on human welfare.) Pollution is assumed to constitute a problem only if associated damages have an economic value. Data limitations and other uncertainties notwithstanding, the damage function approach focuses directly on the link between pollutants and damages, measured in both physical and monetary terms. Thus, of the three approaches discussed here, the damage function approach is the most defensible on the basis of theory. In addition, it is the only method that applies, at least in theory, to all of the damages that might be linked to coal use. For this reason, agencies, including the U.S. Department of Energy and the Federal Energy Regulatory Commission (FERC), rely on the damage function approach for estimating the external costs of electricity production (FERC 1992).

The extent to which firms have already undertaken efforts to internalize some of the external costs resulting from the generation of electricity also has important implications for the socially efficient quantities of electricity and corresponding pollution. The effect of such efforts is to internalize some previously external costs. The value of marginal external costs is based on a specific relationship between the firm's production function and the amount of pollution produced per unit of output. A combination of (1) the amount of pollution produced per unit of output and (2) the value of the damages attributable to each unit of pollution determines the value of external costs at each level of output. For example, if because of government regulation a firm reduces the quantity of pollution it generates, marginal external costs per unit of output are likely to decrease (Just et al. 1982, pp. 274-275). This example would be the case if the types of pollutants produced are unchanged, but the quantity of one or more of the pollutants produced per unit of output is reduced.

If firms reduce pollution in response to regulatory pressures, without knowing the actual value of the marginal external costs associated with electricity production and pollution control, it is not possible to determine whether the resulting level of pollution is efficient. Three outcomes are possible: the efficient level of pollution, under control relative to the efficient level, and overcontrol relative to the efficient level. Freeman et al. (1992) demonstrated that the correct value of any adder should be included when calculating the marginal social costs of electricity. This value depends on the type of policy instrument used.

If pollution has been controlled via direct regulation, any remaining external costs should be added to private costs to ensure that total social costs are accounted for. This fact is true regardless of whether the levels of output and pollution currently produced by the affected firms are efficient. In any event, the pollution that continues to be produced is not factored into the decision-making process and thus must be accounted for by some other means (e.g., adders). While one might be tempted to argue for a negative adder to compensate for overcontrol attributable to excessive restrictions on pollution, this step would only further distort the market.

In contrast, if pollution has been controlled via permits, the correct adder is 0, regardless of whether the current levels of output and pollution are efficient. For permits, as long as the rules that govern trading of permits reflect the characteristics of the pollutant being regulated, all external costs have been internalized as well.[2] Assuming the current outcome is efficient, incorporating an adder would mean double-counting external costs. If the current level of pollution control is less than the efficient level, the appropriate response to the under control of pollution would be to decrease the number of permits to a level consistent with the economically efficient level of pollution control. Where policy leads to the overcontrol of pollution, the appropriate response would be to increase the number of permits to a level consistent with the economically efficient level of pollution control.

For charges, the efficient output level occurs only if the charge equals the marginal external cost at the efficient level of output. Thus, if the efficient solution has been achieved, the charge is an accurate measure of marginal damage (external) costs. Because the charge is paid for each unit of pollution emitted by the firm, external costs have been internalized. Incorporating adders in this case would also mean double-counting external costs. If the current level of pollution control is less than the efficient level, the adder should be set equal to the *difference* between the current charge and the actual external costs. Where policy results in excess pollution control, the adder should again be set equal to the *difference*

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[2] Determining the appropriate trading rules depends primarily on the characteristics of the pollutant in question. See Just et al. (1982, pp. 375-382).

between the current charge and the actual external costs; in the case of overcontrol, this amount is negative.

4.2 REVIEW OF EXTERNALITY COST ESTIMATES

A review of the literature on empirical estimates of externalities associated with the coal fuel cycle focused on recent reports by Ottinger et al. (1991) (hereafter referred to as the Pace report because it was prepared by the Pace University Center for Environmental Legal Studies), Pearce et al. (1992), and Szpunar and Gillette (1992). Each of these reports includes a review of studies that have attempted to estimate various components of the external costs of electricity production. However, because the objectives of each study vary, they also provide different points of view.

The Pace report was completed under contract to the New York State Energy Research and Development Authority and focuses on estimates of the external costs of different electric generation technologies. No effort is made to summarize the external costs associated with the "upstream" phases (e.g., mining, transportation, or waste management) of the coal fuel cycle. Another important characteristic of the Pace report is that it provides estimates of external costs produced via all of the methods described in Section 4.1.2. One of the major distinguishing characteristics of the Pearce et al. (1992) study is its focus on the external costs associated with the entire fuel life cycle, as opposed to simply that portion of the cycle associated with generating electricity. The Szpunar and Gillette (1992) study is of special interest, both because it summarizes existing estimates of external costs attributable to electricity generation and because it attempts to transfer some of those estimates to various Asian countries.

4.2.1 Overview of the Studies

4.2.1.1 Pace Report

The Pace report represents an ambitious undertaking. It begins with an extended discussion of the justification for considering external costs in a policymaking setting. It also discusses theoretical issues related to such questions as the initial allocation of property rights and the corresponding implications for correctly measuring the value of damages attributable to pollution. As noted above, the report limits its discussion of external costs to those associated with fuel use and waste disposal. No effort is made to incorporate external costs associated with up-front activities (e.g., exploration, mining, processing, and transport of fuels). To this extent, the reported values of external costs could be viewed as lower bound estimates. However, other factors offset this source of downward bias.

In general, the impacts (i.e., damages) considered in this report are limited to adverse effects on human health, flora and fauna, materials, and social assets (e.g., recreation and visibility). With respect to the coal-fired generation of electricity, the report gives damage cost estimates for CO_2, SO_2, NO_x, and particulates. (Although SO_2 is a precursor to acid deposition, the effects are treated separately.)

The report also considers acid deposition, three types of solid wastes that result from coal combustion (i.e., bottom ash, fly ash, and sulfur by-products), and possible adverse effects attributable to land and water use by coal-fired generating units. However, the authors could not find any studies of the external costs of these impacts that could be used to produce estimates of the form required for the report (i.e., measured in cents per kilowatt-hour).

Estimates of external costs (i.e., adders) are calculated for individual electric-generating technologies, including four types of coal-fired units — a base unit with no sulfur cleaning equipment, atmospheric fluidized-bed combustion (AFBC), integrated coal gasification combined-cycle (IGCC), and a unit that conforms with New Source Performance Standards. Costs are presented separately for SO_2, NO_x, particulates, and CO_2. The costs for each type of emission are determined by aggregating the available estimates of the different types of damages attributable to each type of emission. This approach contrasts with the Pearce et al. report.

4.2.1.2 Pearce et al. Report

The Pearce et al. (1992) report provides a comprehensive description of the issues related to the external costs associated with fuel cycles. This study is similar to the Pace report in its treatment of the theoretical issues related to the measurement and role of external costs in a social cost framework. However, the Pearce et al. report goes beyond the Pace report because it considers the entire fuel cycle. For example, it considers, at least qualitatively, the external costs associated with each phase of the coal fuel cycle, beginning with mining and continuing to managing combustion wastes. However, as in the Pace report, quantitative estimates are not available for a number of potential external costs. In addition, both reports review many of the same studies in developing representative estimates of specific sources of external costs.

The Pace and Pearce et al. reports differ in important respects. First, the purpose of the Pearce et al. report is to develop representative estimates of the external costs of fuel cycles in the United Kingdom. Thus, the Pearce et al. report reviews many European studies that were not considered in the Pace report. In addition, all cost estimates are measured in pence per kilowatt-hour. For estimates originally reported in U.S. dollars, the conversion is based on exchange rates and inflation estimates reported by The World Bank. This conversion of monetary units could present some difficulties in reconverting the estimates to dollars. In addition, the two reports sometimes use different emissions factors, which has the effect of altering the estimate of monetary damage per kilowatt-hour.

A second major difference between the two reports concerns the manner in which external cost estimates are presented. As noted, the Pace report gives external costs by type of emission for different generating technologies. The individual estimates are then aggregated to form a measure of external cost by generating technology. The Pearce et al.

report gives external costs only by generating technology and disaggregates them by type of external cost (e.g., health effects vs. materials damages). There is no disaggregation according to the type of emission in question, which complicates a side-by-side comparison of the results of the two studies.

4.2.1.3 Szpunar-Gillette Report

The purpose of the Szpunar-Gillette report is to provide a preliminary cost-benefit assessment of the economic impacts of the adoption of clean coal technologies and other pollution mitigation measures related to the coal-fired generation of electricity in various Asian countries, including Indonesia, Thailand, and Taiwan. On the cost side, the assessment focuses on the incremental costs of specific technologies compared with the base case, which consists of a pulverized coal-fired (PC-fired) plant equipped with no SO_2 emissions control and a moderate level of particulate control. The alternative technologies include a PC-fired plant with SO_2 emissions control and an increased level of particulate control, an AFBC plant, a pressurized fluidized-bed combustion plant, and an IGCC plant. Benefits are estimated by applying selected estimates of the value of the external costs of different types of emissions to the projected reduction in emissions attributable to each technology.[3]

The first part of the study briefly reviews the theory of externalities and the rationale for including them in the decision-making process.[4] In addition, the various approaches used to estimate external costs are briefly given. The potential adverse effects attributable to the major types of air emissions associated with the coal-fired generation of electricity are also examined. While the primary source of damages consists of adverse health effects, potential effects on visibility, materials damage, and agriculture are also briefly discussed.

Of specific interest here is the report's review of estimates of external costs for the following pollutants: SO_2, NO_x, TSPs, CO_2, N_2O, CO, VOCs, and CH_4. Values for the external costs attributable to each of these pollutants are reported from as many as nine different sources. Table 14, which has been reproduced from Szpunar and Gillette (1992), summarizes those estimates. One of the more striking facts revealed in Table 14 is the amount of variability in the cost estimates, both within and across pollutants. For example, the estimates of the external cost of 1 ton of NO_x range between \$69 and \$40,000. In

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[3] Recall that the reduction in external costs attributable to a particular action represents the benefits of that action from society's perspective.

[4] It is worth noting that Szpunar and Gillette use the terms "external cost" and "social cost" interchangeably. The usual convention is to use social cost to refer to the sum of private costs and external costs, and this definition is used in this report.

TABLE 14 Sample Values Used by State Public Utilities and Others for Environmental Externalities (1989 $/ton emitted)[a]

Pollutant	Ore.[b]	Mass.[c]	N.Y.[c]	Calif.[c] SDG&E/SCE and PG&E[d]	Nev.[c]	BPA[b]	Other Sources	Pace Study[e]	Lave[f]
SO_2[g]	1,500	1,500 (400)[h,i]	820 (486-669)[j] (832)[l] (2,200)[m]	18,300 and 4,060	1,560	1,500[b]	590- 1,800[k]	4,060	60/990
NO_x	884	6,500	1,780 (2,700-3,417)[j] (1,832)[l]	24,500 and 7,100	6,800	69/884[b]	2,700- 40,000[k]	1,640	70/450
TSP[n]	1,540	4,000	320 (333)[j]	5,300 and 2,380	4,180	167/1,540[b]	--[o]	2,380	100/1,500
CO_2	--	22 (2-6)[i]	2 (18-44)[j] (1)[l]	26 and 26	22	6[p]	15-56[k] 68- 6,480[k]	14	2/10
N_2O[q]	--	3,960	--	--	4,140	--	3,700[k]	--	--
CO	--	860	--	--	920	--	--	--	Negl[r]
VOC[s]	--	5,300	--	17,500 and 3,300	1,180	--	--	--	600/1,800
CH_4[t]	--	220	--	--	220	--	--	--	--

See footnotes on next page.

TABLE 14 (Cont.)

[a] Values are largely based on cost of control; Lave values are based on damage costs.

[b] U.S. Department of Energy (DOE), Bonneville Power Administration (BPA), May 1991; Grahame (1991), October testimony, Attachment 8.

[c] Wiel (1991), p. 50.

[d] SDG&E/SCE = San Diego Gas & Electric/Southern California Edison; PG&E = Pacific Gas & Electric.

[e] Ottinger et al. (1990).

[f] Lave (1991) testimony. The lower value is the best estimate for Massachusetts attainment areas; the higher value reflects scientific uncertainty and estimates possible health effects. Lave uses damage-based values, not cost-of-control values.

[g] After the year 2001 and under emissions trading, if an energy source would offset its SO_2 emissions, economic theory would place its value at zero because the effect would be internalized. If emissions were not offset, the given value would remain.

[h] Between 1996 and 2001, the estimated difference between the external and internal values is $400/ton; after 2001, the estimated difference is zero.

[i] Grahame (1991), October testimony.

[j] New York State Energy Office draft value.

[k] Haites (1990).

[l] Grahame (1991), October testimony, Attachment 7.

[m] Draft New York State Energy Plan 1991; Grahame (1991), December testimony.

[n] TSP = total suspended particulates.

[o] -- = not calculated.

[p] DOE, BPA, February 21, 1991; Grahame (1991), October testimony, Attachment 2.

[q] N_2O = nitrous oxide.

[r] Negl = calculated but not significant.

[s] VOC = volatile organic compounds.

[t] CH_4 = methane.

Source: Szpunar and Gillette (1992).

addition, estimates of the external cost of 1 ton of CO_2 range as low as $2, while the same estimates range as high as $17,500 for VOCs, $18,300 for SO_2, and $40,000 for NO_x. The estimates presented in the Pace report generally lie in the middle of these ranges.

4.2.1.4 Summary

Together, the three reports reviewed in this section comprehensively summarize the extant literature on externalities associated with the coal fuel cycle. As such, the estimates presented provide a good picture of what is currently known about the magnitude of certain external costs. Nonetheless, a number of important factors must be considered before reviewing specific estimates. First, in certain cases, the external costs presented are based on control costs, rather than on some measure of the damages that result from the adverse impacts in question (especially in the Pace report). Control costs are not a legitimate measure of damage costs (Section 4.1.3.1). Consequently, external costs measured via this method are useless for the purposes they were intended to serve.

Second, estimates are not available for many of the potential external costs associated with the coal fuel cycle. On the basis of the results summarized in the reports reviewed here, the estimates of relevant external costs available are limited to the use of coal in generating electricity. Estimates of the external costs attributable specifically to the use of coal in the residential and commercial sectors or in other industrial uses are not readily available. Furthermore, the authors of the reports reviewed here were unable to locate any estimates of the external costs associated with land or water impacts that result from the use of coal. They were also unable to locate estimates of the external costs associated with most of the phases of the coal fuel cycle (e.g., mining activities, coal transportation, and waste management).

Third, considerable disparity is found in the estimates of currently available external costs. Part of the variation among the various estimates of the external costs attributable to a particular pollutant or type of damage is explained by the following:

- Differences in the characteristics of the affected environment,
- Differences in estimation techniques and the state of the art at the time a particular study was completed,
- Different assumptions regarding emission rates per kilowatt-hour for specific pollutants, and
- "Professional judgment" by the researchers involved in each of the studies.

The bottom line is that it is not possible to meaningfully assess the validity of the individual estimates or state which of the estimates presented below is most valid.

4.2.2 Summary of Available Estimates

Comparison of the cost estimates included in each of the three reports is complicated by a number of factors. First, the Pace report presents cost estimates for each type of emission in the form of dollars per pound of emissions. These estimates are then combined with assumptions about emissions rates for different generating technologies and aggregated to obtain an estimate of the external costs, measured in cents per kilowatt-hour for each generating technology. In the Pearce et al. report, external costs are reported by generating technology for each type of damage rather than by type of emission. Moreover, because the Pearce et al. report is based on the situation in the United Kingdom, cost estimates are reported in pence per kilowatt-hour. This method of computing costs complicates efforts to directly compare the estimates in the Pearce et al. report with those in either the Pace report or the Szpunar-Gillette report.[5] Finally, the estimates presented in the Szpunar-Gillette report, including the estimates from the Pace report, are measured in dollars per ton of pollutant. Once again, comparison with the figures in the Pearce et al. report is complicated by the manner in which the estimates in that report were constructed.

4.2.2.1 Air Quality Impacts

The external cost estimates considered in the three reports are confined to those costs attributable to certain types of air emissions and specific types of damages. In the Pace and Pearce et al. reports, the air emissions for which external cost estimates are reported include SO_2, NO_x, particulates, and CO_2. The damages attributable to the emissions for which cost estimates are reported include health effects (both mortality and morbidity), crop losses, forest degradation, materials damages, global warming, and decreased visibility. In addition to the four types of emissions just noted, the Szpunar-Gillette study also reports external cost estimates for N_2O, CO, VOCs, and CH_4. The estimated external costs of these additional types of emissions generally consist of values reportedly used by selected states — Massachusetts and Nevada — in the process of constructing externality adders. Insofar as the validity of these estimates is concerned, no indication is given of the methods used to derive these figures. Moreover, the fact that these cost estimates were not discussed in either of the other two reports suggests that they have been derived via ad hoc methods. Thus, they are not considered further in this report.

The damages attributable to SO_2 for which estimates are provided include health effects, forest degradation, materials damages, and visibility effects. Damages attributed to NO_x include health effects, crop losses, materials damages, and visibility impacts. Damage costs attributable to particulates include health effects and visibility impacts. The estimated external costs attributable to these three types of emissions for the case of a coal-fired unit without SO_2 scrubbers are presented in Table 15.

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[5] This complication occurs because it is not clear what exchange rate should be used to convert pence to cents. In many cases, the values reported had to be converted from other currencies to the British measure. Thus, an additional problem is posed by successive conversions based on potentially inconsistent exchange rates.

TABLE 15 Comparison of External Cost Estimates by Type of Damage for SO_2, NO_x, and Particulates: Coal-Fired Units without Scrubbers

	Pace Report	Pearce et al. Report	
Type of Damage	Cents/kWh[a]	Pence/kWh[b]	Cents/kWh[c]
Health effects			
Mortality	3.48	0.32	0.54
Morbidity	0.28	0.12	0.34
Crop losses	0.006	0.10	0.17
Forest	0.00	0.84	1.43
Materials	0.23	3.22	5.47
Visibility	0.50	0.00	0.00
Total	4.50	4.60	7.95

[a] Values are based on Table 1, Chapter VI, Ottinger et al. (1991). The values given in the Pace report are expressed in terms of 1989 prices. These values were inflated to 1990 prices (inflation rate = 4%) to make them more comparable to the values reported in the Pearce et al. report.

[b] Values are taken from Table 19.1, Pearce et al. (1992). These values are expressed in 1990 prices.

[c] Pence per kilowatt-hour were converted to cents per kilowatt-hour by using the exchange rate of 1 British pound sterling = $1.7. This exchange rate was apparently used by Pearce et al.

As the data in Table 15 indicate, the cost estimates reported by Pearce et al. exceed those in the Pace report by approximately 77%. This difference is generally attributable to a much larger estimate of the external costs of materials damages by Pearce et al. This difference results from the fact that the Pearce et al. report included estimates of damages attributable to acid deposition. The Pace report did not include damage estimates attributable to acid deposition, arguing that defensible estimates were not currently available. This divergence in opinions on the availability of usable estimates of such costs is partly explained by the damage estimates for acid deposition included in the Pearce et al. report. These estimates were constructed for locations in Scandinavia and other parts of Europe.

The difference in the estimates of materials damages is somewhat offset by a higher estimate of health-related damages, as well as the inclusion of visibility impacts in the Pace report.

Damages resulting from CO_2 emissions are linked to the adverse effects of global warming. In the Pearce et al. report, damage costs were estimated by extrapolating damage estimates from a study by Cline (1992), as modified by Pearce et al., to estimate costs per unit of CO_2. In the Pace report, damage costs were based on estimates of the costs of carbon sequestration, a mitigation strategy for which a variety of cost estimates have been produced. As such, these estimates are not comparable. Furthermore, because no relationship exists between damage costs and control costs, the estimate presented in the Pace report is of no value as a legitimate measure of external costs.

4.2.2.2 Water Quality Impacts

All three studies recognize the potential for external costs to arise from water usage in the coal fuel cycle. The Pace and Pearce et al. reports discuss in detail the possible adverse impacts and corresponding external costs of effects from cooling water requirements (including impingement and entrainment of aquatic organisms), the effects of acid deposition in lakes and streams, and possible groundwater contamination from coal storage and management of coal wastes. However, both studies concluded that currently available studies do not include sufficient detail to construct empirical estimates of any external costs that might arise.

4.2.2.3 Land Impacts

The Pace and Pearce et al. reports recognize various adverse land-related impacts attributable to the coal fuel cycle. However, both studies concluded that currently available studies do not include sufficient detail to construct empirical estimates of any external costs that might arise.

4.3 TRANSFERABILITY OF ESTIMATES OF EXTERNAL COSTS

Given the current availability of estimates of the external costs attributable to various air emissions from electric utilities, the question arises whether and to what extent these estimates could be transferred to other situations. Two cases are of particular interest:

- To what extent could the external costs associated with the coal-fired generation of electricity be transferred to other production processes that involve coal (e.g., residential home heating and industrial usage)?
- Could these estimates be transferred to other locations, and to what extent could they be transferred?

Depending on the situation, one or both of these questions must be answered. However, answering either question involves essentially the same considerations. Because the discussion that follows applies equally to both situations, the distinction between the two situations is ignored for the remainder of this report.

To see why transferability is important, recall that external costs are estimated on the basis of specific situations. For example, in the damage function approach, the extent of physical damages attributable to a pollutant depends on a number of factors, including atmospheric conditions, the composition of the natural environment, and the extent to which the affected human population is susceptible to the types of damages caused by the particular pollutant (Section 4.2.1.3). Damages must then be converted to monetary units by means of one or more currently available valuation methods. As discussed below, monetary values are also subject to various situation-specific influences. Likewise, external costs based on mitigation costs depend on, among other things, the prices used to determine mitigation costs. These prices are, in turn, a function of current market conditions. All of these factors must be considered in determining whether existing estimates can be applied to new and different situations.

As noted in Section 4.1.2.2, control costs are not related to damage costs in any systematic fashion. On the basis of current U.S. policy, the equality of marginal control costs and marginal damage costs is, at most, pure coincidence. Thus, the transferability of control costs is not considered here. Control costs incurred by coal-fired electric generating units in the United States might be transferable to other locations as a means of estimating the costs of pollution control that might be incurred elsewhere. However, this issue is not addressed in this report.

4.3.1 Determinants of the Validity of Benefits Transfers

Recalling the discussion in Section 4.1, the benefits of a reduction in pollution are measured by the concomitant reduction in damage cost. Thus, the transfer of damage cost estimates and "benefits transfer" amount to the same thing. Benefits transfer has received increased attention in recent years. This attention largely reflects the growing need for estimates of the monetary value of the benefits of improvements in environmental quality and the considerable cost, in time and money, that is almost always incurred in completing such studies. To the extent that benefits transfer is a viable alternative to estimates based on primary data, significant savings could be realized.

The issues related to benefits transfer are discussed in a number of recent sources, including the Pearce et al. report and a set of articles in Desvousges et al. (1992). For clarity, the discussion that follows adopts terminology developed in the context of benefits transfer (Desvousges et al. 1992). The term "study site" refers to a location (situation) for which damage costs have already been estimated by using primary data. The term "policy site" refers to the location (situation) for which estimates of damage costs are sought.

Boyle and Bergstrom (1992) (p. 659) have proposed the following set of "idealistic" technical criteria that must be met for a benefits transfer to be successful:

> ...1) the nonmarket commodity valued at the study site must be identical to the nonmarket commodity to be valued at the policy site, 2) the populations affected by the nonmarket commodity at the study site and the policy site

have identical characteristics, and 3) the assignment of property rights at both sites must lead to the same theoretically appropriate welfare measure (e.g., willingness to pay versus willingness to accept compensation).[6]

It is important to emphasize that, as Boyle and Bergstrom note, these are idealistic criteria. It is unlikely that each of these criteria will be met in practice. Nonetheless, they provide a benchmark for assessing the potential for successful benefits transfer in a specific situation.

The Pearce et al. report also addresses the issue of benefits transfer. Following a discussion of the types of transfers that might be attempted, the report identifies various issues likely to be encountered in benefits transfers. These issues, many of which overlap or coincide with those identified in other studies, include determining what is to be transferred, comparability of the study site and policy site, the effects of aggregation on the potential for bias in resulting benefits estimates, and the relationship between average and marginal measures of benefits (damage costs).

4.3.1.1 Types of Transfers

First, it must be determined what is to be transferred. In its most direct form, benefits transfer could consist of using unit values (e.g., the value of a user-day of fishing) or the health damages of exposure to a particular concentration of pollutant *X*, derived for a study site to estimate the damages (external costs) at a policy site. However, the validity of this approach significantly depends on the comparability of the two locations in a number of dimensions.

A second approach would be to transfer the *functions* used to estimate damages. Depending on the specifics of a particular situation, this approach may involve the transfer of one or a set of functions. For example, a policy might be predicted to improve the quality of recreation at a site. If a function for recreation has been estimated for another site, this function might be transferred to the policy site. The resulting function would be used to estimate the change in demand (quantity) for the site resulting from the proposed policy as well as the economic value of that change. In other situations, it might be necessary to transfer both a damage function and a benefits function (which estimates the monetary value of damages) to the policy site.

To illustrate this approach to benefits transfer, consider the following example. Mitigation costs consist of the expenditures required to offset the adverse effects of the

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6 The last of these three conditions has to do with the fact that, depending on the initial assignment of property rights, someone affected by a specific change may have to pay, or be compensated for, the change to occur. For example, if A wants an improvement in environmental quality, but this request would require B to give up something he/she owns (i.e., for which he/she has the property right), the correct measure of the benefits realized by A is the amount he/she would be willing to pay. Alternatively, from B's perspective, the value of what is given up by B is measured by the amount of compensation he/she would require in order to be as economically sound as he/she was before the change.

pollutant in question. As such, these costs depend on both the amount of inputs that must be purchased and the market price of those inputs. On the basis of market conditions, such costs could vary widely by location. Consider again the adverse effects on agriculture. Assume that a particular type of emission causes average yields of affected farms to fall by 10%. This decrease in yields might be offset by using more fertilizer, water, or land (or some combination of these or other inputs). Assume, for expositional convenience, that the only option is to increase the amount of land in the production process. The additional land required depends on the productivity of the land and the additional output required to offset the 10% reduction attributable to pollution. The mitigation costs are then equal to the quantity of land times its per unit price plus any variable costs (e.g., seed, fertilizer, and labor) required to produce the additional output.

The resulting additional costs could be related to emissions and output of electricity by dividing the increased costs per time period by the emissions attributable to electricity production and the total output of electricity per time period (measured in kilowatt-hours), respectively. To transfer this estimate to another location, it would be necessary to assess the comparability of agricultural productivity and production costs at the two locations, (i.e., and to what extent at the study site and the policy site). For example, if land is more productive at the study site than at the policy site, and if all factors remain the same, a direct transfer of the estimated costs would understate the costs at the policy site. The reason is that additional land (and variable inputs) would be required to fully offset the reduction in yields at the policy site. Comparability of input and output prices at the two locations would also have to be assessed.

4.3.1.2 Comparability of the Policy and Study Sites

The validity of either of the types of benefits transfer (i.e., the transfer of unit values or the transfer of damage functions) is greatly influenced by the comparability of the affected populations. Comparability must be assessed along two dimensions: physical and socioeconomic. The physical dimension includes the characteristics of both the physical environment and the human, animal, and plant populations. The socioeconomic dimension is concerned with factors that influence willingness to pay (or willingness to accept compensation) for changes in environmental quality and concomitant changes in human health and welfare.

The characteristics of the physical environment (e.g., ambient concentrations of chemicals, pollutants, and biological oxygen demand) determine the impact of a marginal change in some characteristic such as the level of pollutant X on the level of environmental quality. At the same time, the characteristics of the affected population are important determinants of the susceptibility (and potential damages) attributable to different pollutant concentrations. For example, the old and the very young (infants and adolescents) are more likely to be adversely affected by certain pollutants than are young and middle-aged adults in good health. Such characteristics are factored, either implicitly or explicitly, into the estimate of the parameters of a damage function that relates a specific type of damage (e.g., adverse health effects) to a particular pollutant. Thus, a significant difference in the baseline conditions that characterize the physical environment at the two sites may seriously undermine the validity of a benefits transfer.

The monetary value of a change in environmental damages depends on the amount the affected population is willing to pay. In most textbook treatments of the theory of demand, determinants of willingness to pay include the quantity of the good, income, tastes and preferences, prices of related goods, and expectations about the future. Willingness to pay is generally assumed to be positively related to income. Thus, for example, if the average income of the population at the policy site is half of the average income of the population at the study site, it could be argued that, all else being constant, the value of damages at the policy site would be less than the value of the same quantity of damage at the study site.

The role of tastes and preferences is especially relevant in the context of this report. Specifically, it is reasonable to expect that the willingness to pay for environmental improvements is influenced by the potential trade-offs between such improvements and other goals and objectives. Efforts to reduce pollution (e.g., emissions from coal-fired facilities), generally increase the costs of production in terms of money and resources. Such increased costs could translate into reduced rates of economic growth and economic well-being. Depending on an economy's current standard of living, such trade-offs may be unacceptable or at least very costly. All else constant, the result is likely to translate into a decreased level of willingness to pay for reductions in pollution damages in developing countries relative to willingness to pay in developed countries. As a consequence, transferring willingness-to-pay estimates from the United States to China could overstate external costs.

4.3.1.3 Aggregation

Another important issue in the context of benefits transfer concerns the aggregation of different damage estimates. As discussed in Cantor (1991) and summarized in the Pearce et al. report, this issue involves numerous different aspects. However, it is important to note that this issue will arise regardless of whether the analyst uses benefits transfer or primary studies to estimate the external costs of a particular activity such as the coal-fired generation of electricity. Nonetheless, this issue is addressed to highlight its implications for the validity of the estimates of external costs.

As discussed in the Pearce et al. report, problems can arise with respect to aggregation across the same endpoints (i.e., same type of damage) at different locations and across different endpoints (i.e., different types of damages). In the former, if two or more sites are substitutes for each other, as might occur in the case of recreation, totaling the value of the change in the same type of damages at each site overstates the value of the change in damages if the two sites are considered simultaneously. In the latter, if the affected goods are substitutes, aggregation of individual damages again overstates damages. However, if the two goods are complements, simple aggregation underestimates damages (if no changes occur).

4.3.1.4 Average vs. Marginal Values

The issue of average versus marginal values concerns the type of value estimated at the study site and the type of change to be evaluated at the policy site. Most studies of external costs consider the value of damages as a function of the ambient concentration of a particular pollutant. Thus, damages expressed in terms of dollars per unit of a pollutant are a measure of the average damages attributable to the pollutant (Pearce et al. 1992). However, when a particular source of emissions is being considered, it is often more appropriate to consider the marginal impact of the emissions in question. This case is especially true when multiple sources of the same type of emission are in the region, or when the policy in question would change total emissions by a relatively small proportion of the total.

Mathematically, a well-defined relationship exists between average and marginal values. Specifically, as long as the marginal value of the n'th unit is less than the average value of the n units, a small change in quantity will cause the average to decline. On the other hand, if the marginal value of the n'th unit exceeds the average value associated with the n units, a small change will cause the average to increase. Thus, depending on the shape of the average value function, using a change in average values to measure the value of the incremental change in question could either understate or overstate the value of the marginal change. For example, if the average damage costs associated with emission X decrease over the relevant range (i.e., the amount of emissions currently being generated), using the value of the change in the average damages resulting from a small decrease in X would overstate the value of the actual change in damages. If average damage costs increase over the relevant range of emissions, using the value of the change in the average damages resulting from a small decrease in X would understate the value of the actual change in damages.

4.3.2 Summary

Clearly, the question of whether and to what extent externality values can be transferred from one situation to another involves complex issues. At best, it appears that benefits transfers could help define the potential magnitude of social costs of the fuel cycle as it relates to coal-fired electricity generation. However, even in this case, a number of sources of external costs have not been addressed systematically. Attempts to transfer such values from one country to another appear to be highly suspect. This conclusion is based not only on the potential differences with respect to the physical damages associated with specific pollutants and actions, but on the likely differences in willingness to pay in the two countries to reduce or avoid such damages.

With respect to transferability of the externality values reviewed here to the situation in China, a number of points are worth noting. If only the relative lack of pollution control in China combined with the close proximity of pollution sources to high-population areas were considered, it might be reasonable to conclude that the costs reviewed here would constitute a lower bound on the damage costs incurred in China. However, this conclusion overlooks the fact that significant differences may exist in the willingness to pay for pollution control in the United States and China, which could push the bias in the opposite direction. To reduce pollution, resources must be reallocated from other productive activities. Reallocation could then adversely affect economic growth. In a developing economy, such as in China, this trade-off may be viewed as more costly than it is in the United States, where the standard of living is much higher. In conclusion, any effort to transfer externality cost estimates to the Chinese situation would be highly suspect.

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