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INTEGRATING LOCAL, REGIONAL AND GLOBAL ASSESSMENT IN CHINA'S AIR POLLUTION CONTROL POLICY

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CIES POLICY DISCUSSION PAPER 0023

INTEGRATING LOCAL, REGIONAL AND GLOBAL ASSESSMENT IN CHINA'S AIR POLLUTION CONTROL POLICY *

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ABSTRACT

This paper develops a consistent analytical framework to assess the environmental damage of air pollution in China. Spatial models are used to estimate the impact of air pollution on human health, ecosystems, and global climate change. While the health effect is largely borne by local residents, the environmental impact of China's air pollution goes beyond its borders. Sulfur emissions from the combustion of coal cause acid rain, which falls in neighboring countries and harms the ecosystems in the East Asian region. At the global level, emissions of carbon dioxide from the burning of fossil fuels enhance the greenhouse effect and contribute to global warming. A combination of willingness to pay and marginal social cost approaches are used to evaluate these environmental damages. Alternative abatement scenarios are explored to identify optimal control strategies that are based on overall cost-benefit analysis of emission abatement at multiple levels. The results suggest that the largest source of damage by air pollution in China is human mortality and morbidity associated with ambient concentrations of fine particulates and sulfur dioxide. Improving urban air quality will deliver substantial benefits. There is also considerable synergy between local, regional and global abatement by adopting measures that target the sources of air pollutants.

Keywords: air pollution, human health, ecosystems, global climate change, China

JEL codes: Q25, I18, P20

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NON TECHNICAL SUMMARY

China has been one of the fastest growing economies in the world for over two decades since the launch of economic reforms in 1978. Accompanying and fostering China's economic growth are its energy demand increases reflecting industrialization, urbanization, and increased residential energy use to sustain a higher standard of living. The increase of energy consumption, particularly the dominance of coal in China's fossil fuels has resulted in severe air pollution. Ambient concentrations of small particulates and sulfur dioxide – both are byproducts of coal consumption – are among the world's highest. Such high concentrations of ground-level air pollutants impose grave dangers to human health for million of urban residents. While the health risks are borne by local residents, the environmental impact of China's air pollution goes beyond its borders. Sulfur emissions from the combustion of coal cause acid rain, which may be carried by wind to fall in neighboring countries such as Japan and Korean, damaging crops, forests and fisheries, harming the ecosystems in the East Asian region. At the global level, emissions of carbon dioxide by China from the burning of fossil fuels enhance the greenhouse effect and contribute to global warming.

This study integrates the local, regional, and global assessment of China's air pollution control policy in a consistent analytical framework. Spatial models are used to evaluate the impact of air pollution on human health, ecological damage and global climate change. Alternative abatement scenarios are explored to identify optimal control strategies that are based on overall cost-benefit analysis of emission abatement at multiple levels. Abatement of urban air pollution, for example, given its adverse health impact, should primarily be given priority in China. But locally motivated air quality programs such as one that only promotes the use of electrostatic precipitators, while effective in reducing particulate emissions, will have limited benefits in terms of reducing acid rain and protecting the global climate; whereas measures targeting the sources of pollution, such as improving energy efficiency to reduce the use of fossil fuels, can achieve synergy in reducing emissions of particulates as well as sulfur dioxide and carbon dioxide from fossil fuels.

The starting point of this inquiry is the analysis of China's acid rain abatement scenarios in the World Bank's recent *China 2020* study. The aim is to extend the acid rain scenarios which focused on regional abatement strategies to considering both local and global implications. Three types of air pollution are jointly assessed: ambient concentration of small particulates ($PM_{2.5}$) in urban areas, sulfur emissions (SO_2) which result in regional acid deposition, and carbon dioxide emissions (CO_2) which contribute to global warming. Modeling projections in this study show that, in the absence of abatement actions, ambient concentration of particulates and annual emissions of sulfur and carbon will more than double between 1990 and 2020.

A clear conclusion of the analysis in this paper is that the largest source of damage by air pollution in China is human mortality and morbidity associated with ambient concentrations of fine particulates and sulfur dioxide. Adoption of abatement measures to improve urban air quality will have large benefits. The analysis also indicates that there is considerable synergy between local, regional and global abatement. The sources of emissions for the three pollutants - particulate, sulfur dioxide and carbon dioxide - are all by-products of fossil fuels, especially coal. Control policies that target the sources of these air pollutants will not only benefit the health of Chinese residents, but also reduce acid rain in East Asia and address the global warming problem.

Among the abatement scenarios examined, improving energy efficiency, especially increasing the efficiency of coal use appears to be the most cost effective. This is extremely attractive as coal is the main source of air pollution at the local, regional and global level, and China will only gradually reduce its reliance on coal. By increasing efficiency and reducing energy consumption, the intensive energy efficiency scenario targets at the source of the pollutants emitted by fossil fuels.

Economy-wide analysis indicates that improving energy efficiency also entails substantial indirect benefits flowing on to the rest of the economy through increased productivity in energy end-user sectors. In addition to technical improvement, increasing energy efficiency must emphasize removing distortions and breaking down market barriers. Welfare decomposition reveals that the benefits of an energy input-augmenting technical change can be significantly offset by policy distortions such as export subsidies or import tariffs.

Despite China's achievement in reducing energy intensity at a rate 5% annually in the past two decades, there appears to be scope for further improvement given the gap in energy intensity between China and other countries. China's current energy intensity more than doubles that of other Asian developing countries. Many projects that would save substantial energy and yield sound returns remain to be implemented, particularly in setting up information systems and strengthening energy management institutions. Internalizing the externalities of environmental pollution by increasing prices of fossil fuels to reflect their true social costs is critical in accelerating efficiency improvement.

Improving energy efficiency by itself will be insufficient to prevent further large increase in China's emissions. A more complete strategy will have to involve larger emission reductions, particularly through targeting small emission sources and critical areas. Adopting abatement measures such as coal-washing, fluidized bed combustion, and sulfur scrubbing are effective in supplementing energy efficiency improvement to reduce emissions. Also, transporting higher quality coal from the surplus producing northern region to the southwest and eastern coastal regions where local coal is high in sulfur can be an attractive option. The challenge is to transport coal over long distances, which adds to the already great pressures on the transport system. Reforming the transport sector to use market to determine the prices of transport services, and investing in transport infrastructure are integral elements of China’s long term energy strategy.

The scope of fuel substitution into natural gas and renewable energy (including hydroelectric power) appears to be limited, and the costs involved in building these energy infrastructures are high. The key challenge will be to develop a sound government regulatory framework to encourage greater private sector participation and attract investment from international lending organizations to mitigate the risks.

The findings in this study suggest that an optimal air pollution control strategy needs to integrate all the abatement measures at the local, regional and global level, and also to take into account of feedback from the economy. While single-purpose economic instruments can be an appropriate response for one pollutant, more complex policy packages targeting pollution abatement at multiple levels and including measures such as open trade policy will provide a better outcome than any individual action.

INTEGRATING LOCAL, REGIONAL AND GLOBAL ASSESSMENT IN CHINA'S AIR POLLUTION CONTROL POLICY [*]

Chao Yang Peng

INTRODUCTION

Rapid economic growth in China poses great environmental challenge. The increase of energy consumption, particularly the dominance of coal in China's fossil fuels has resulted in severe environmental degradation. It was estimated that China incurred at least $54 billion, or nearly 8% of GDP equivalent in environmental damage in 1995 (World Bank 1997). While a local source of damage is urban air pollution that affects the health of millions of city residents, the environmental impact goes beyond China's borders. Sulfur emissions from the combustion of coal, for example, cause acid rain which falls in neighboring countries and harms the ecosystems in the East Asian region. At the global level, emissions of carbon dioxide from the burning of fossil fuels enhance the greenhouse effect and contribute to global warming.

A wide range of abatement measures is available to tackle these environmental problems. Policy responses targeting at each problem in isolation, however, can fall into the trap of a fragmented approach, with competing or inconsistent policies and attrition between agencies with different mandates. Local air pollution, for example, given its adverse health impact, should primarily be given priority in China. But locally motivated air quality programs such as one that only promotes the use of electrostatic precipitators, while effective in reducing particulate emissions, will have limited benefits in terms of reducing acid rain and protecting the global climate; whereas measures targeting the sources of pollution, such as improving energy efficiency to reduce the use of fossil fuels, can achieve synergy in reducing emissions of particulate as well as sulfur dioxide and carbon dioxide emitted from fossil fuels.

This paper attempts to integrate local, regional and global air pollution assessment in China in a consistent analytical framework. The starting point of this inquiry is the analysis of China's acid rain abatement scenarios in a recent study of the

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World Bank's (1997) *China 2020* series. The aim is to extend the acid rain scenarios which focused on regional abatement strategies to considering both local and global implications. Three types of air pollution are jointly assessed: ambient concentration of small particulates ($PM_{2.5}$) in urban areas, sulfur emissions (SO_2) which result in regional acid deposition, and carbon dioxide emissions (CO_2) which contribute to global warming. The objective is to identify optimal control strategies that are based on overall cost/benefit analysis of emission abatement across all three air pollutants.

To do this, three analytical tools are used. The centerpiece of the tools is the Regional Air Pollution Information System for Asia (RAINS-Asia) model which has been developed specifically to investigate acid deposition and ambient air quality in Asia (Foell et al. 1995). A total of 23 countries and 94 sub-regions in Asia are represented in RAINS-Asia. China in the model is divided into 28 sub-regions (including Hong Kong), among which 10 are large metropolitan areas. While RAINS-Asia provides the capability to examine the effects of emission control strategies at a regional scale, it does not allow for accurate estimation of sources of emissions and air quality *within* an urban area, nor does it capture the global impacts of emission abatement options. To address the local and global aspects of China's air pollution problem, two other models are used in tandem with RAINS-Asia. One is the Urban Branching Atmospheric Trajectory (UR-BAT) model which can be embedded in the RAINS-Asia framework. Pollution data generated by RAINS-Asia can be imported to UR-BAT and accurately represented within urban areas at fine resolution (Calori and Carmichael 1998). The other is the Global Trade Analysis Project (GTAP) model which is a global general equilibrium model capable of assessing policy implications of pollution abatement options for China and its trade partners (Hertel 1997). RAINS-Asia projections can be used to inform GTAP which in turn can provide economy-wide feedback.

The analytical framework is presented in Figure 1. Air pollution abatement scenarios extended from those proposed in the World Bank's (1997) *China 2020* study are developed in consultation with the State Environmental Protection Administration (SEPA) in China to make the analysis as policy relevant as possible. These abatement scenarios are first assessed through simulations in the RAINS-Asia model. The results generated are fed into the UR-BAT and GTAP models. Emissions of the three

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and database.

pollutants identified and the associated environmental impacts are assessed. The costs associated with each scenario are then compared to the economic benefits of emission reduction across all three pollutants.

A key part of the analysis is the estimation of economic benefits. For local air pollution reduction, the focus of economic benefits is on health effects. A number of existing studies have estimated the mortality and morbidity rates associated with air pollution in a number of Chinese cities (Xu and Gao et al. 1994, Xu and Johnson 1997, Xu and Yu et al. 1997). The dose-response functions documented in these studies are used to estimate the value of lives saved and the opportunity cost of illness for air pollution abatement in China. The economic benefits of regional acid deposition reduction are derived by estimating the avoided costs of crop yield losses based on results of ecosystem damages generated by RAINS-Asia. The economic benefits of carbon emission reduction are estimated using available valuation in terms of dollars per ton of carbon emissions. In addition, indirect economic benefits of abatement measures such as gains through energy efficiency improvement are estimated using the GTAP model.

This paper begins by discussing the key assumptions used in this study, based on the past experience and future projections of economic growth and energy consumption in China. Damages of air pollution caused by fossil fuel emissions are then assessed. Is the greatest source of damage from urban air pollution, acid rain, or global warming? To what extent they compare to each other? Clarification of these questions leads to the exploration of cost-effective pollution abatement scenarios, using the RAINS-Asia model in connection with UR-BAT and GTAP simulations. Results are presented regarding abatement costs and economic benefits of air pollution controls at the local, regional and global level. This is followed by a brief discussion of policy implications.

MODELING ASSUMPTIONS

China has been one of the fastest growing economies in the world for over two decades since the launch of economic reforms in 1978. Accompanying and fostering China's economic growth are its energy demand increases reflecting industrialization, urbanization, and increased residential energy use to sustain a higher standard of living. China is currently the third largest energy consuming country in the world behind the USA and Russia. On a per capita basis, however, China ranks substantially lower than industrialized countries.

Table 1 shows that China's per capita energy consumption is less than a half of the world's average. Importantly, the table shows that per capita energy consumption increases from low income South Asian economies, through higher income East Asian economies, to industrialized countries in the region, revealing a clear relationship between per capita energy consumption and per capita GDP. As China strives to raise its standard of living, its per capita energy consumption is expected to increase. In particular, annual coal consumption is expected to increase. China relies heavily on coal which accounts for over 75% of its current total primary energy, compared with 25% in the USA and a world average of 27%.

The environmental impact of China's energy consumption is overwhelming. In terms of air pollution, ambient concentrations of sulfate (small particulates) and sulfur dioxide–both are byproducts of coal consumption–are among the world's highest. In 1995, all but two of China's 87 cities monitored for total suspended particulates and more than half of the 88 cities monitored for SO_2 far exceeded the World Health Organization (WHO) guidelines. Some cities such as Taiyuan and Lanzou had SO_2 levels almost 10 times the WHO guidelines (World Resources 1998-99, p. 117). Industry accounts for two thirds of China's coal use. Industrial boilers alone consume 30% of China's coal. These boilers contributing to much of China's ground-level air pollution, especially small particulates and SO_2.

The extent of China's air pollution is more than local. Geographic proximity with neighboring countries makes some of China's environmental problems to be regional concerns in East Asia. Airborne acidic pollutants generated from the combustion of coal in China, for example, are transported by wind to Japan and the Korean Peninsula. Also, industrial emissions of carbon dioxide from China currently ranks the second highest after the USA, contributing to the greenhouse effect and global warming.

Options to reduce China's air pollution are assessed against a baseline where China's future energy consumption pathway is delineated by the assumptions presented in Table 2. It is assumed that China's economy will grow at a pace slower than the double-digit rates as achieved toward the mid-1990s, attaining an average annual growth rate in real GDP at 8.8% from 1990 to 2000, and moving on to a stable growth path in the range of 6.4% and 5.5% over 2000-2010 and 2010-2020, respectively. Population growth in China is expected to decline over the projection period. The annual growth rate halves from 1.2% in 1990-2000 to 0.6% in 2010-2020, with an average rate of 0.91% for the entire period of 1990-2020. Total population is projected to increase from 1.171 billion in 1990 to 1.535 billion in 2020.

The real GDP and population growth assumptions are combined with energy intensities to generate energy consumption path for China. Two sets of energy intensities are considered in RAINS-Asia to project China's future energy consumption:

- Base case. The energy intensities in the base case reflect the hypothesis that no additional measures will be adopted beyond the existing energy policies. Energy use follows the "business-as-usual" practice without strong promotion of energy efficiency measures or fuel substitution to reduce emissions from energy consumption.
- Efficiency case. The energy intensities in the efficiency case assume that China achieves "reasonable" improvements in energy efficiency as the Chinese economy grows. This follows the experiences of industrialized countries during the period 1973-1983, including the response to the increase in energy prices, general improvement in technology, and the increased use of renewable resources.

The difference of the two cases is reflected in the slower increase of energy intensity per capita, but faster decline of energy intensity per unit of GDP in the efficiency case than those in the base case. The resulting energy consumption paths are presented in terms of annual growth in China's primary energy consumption. The base case will require much higher growth in energy consumption to sustain the real GDP and population growth than the efficiency case.

How do these two energy growth paths compare to China's earlier experience in energy use? In the past two decades, China achieved fast economic growth with relatively modest growth in energy needs. Between 1978 and 1997, China's energy demand grew at under 5% per annum, about half of its real GDP annual growth rate of 10%. Despite an equivalent 5% decline in energy intensity per year over this period, China's current energy intensity is still considerably higher than those in peer developing countries and significantly higher than those in industrialized countries (Peng 1996, P. 12). There appears to be scope for further improvement in China's energy intensity. While the base case is less optimistic in this regard, the efficiency case assumes a continuing trend in energy efficiency improvement, that the growth of primary energy consumption is projected to be about half the rate of China's real GDP growth between 2000 and 2020, similar to the earlier rate of energy demand growth relative to GDP growth during 1978 and 1997.

SOURCES OF DAMAGE BY AIR POLLUTION

Local Health Damage

Air pollutants involve a complex mixture of small and large particles of varying origin and chemical composition. Small particles–less than 2.5 microns in diameter–generally come from combustion of fossil fuels. The most common fine particles are sulfates from coal-fired industrial boilers and power plants. In addition to sulfate, SO_2 concentrations are formed when SO_2 emissions from coal condense in the atmosphere. Large particles–from 2.5 to 100 microns in diameter–usually comprise smoke and dust from industrial processes, agriculture, construction, and road traffic.

The health effects of particulates are strongly linked to particle size. Scientific studies suggest that small particles ($PM_{2.5}$) are likely to be most dangerous, because they can be inhaled deeply into the lungs, settling in areas where the body's natural clearance mechanisms cannot remove them. The constituents in small particulates also tend to be more chemically active. As monitoring methods and data analysis have become more sophisticated, the focus of attention has increasingly shifted to fine particulates. The local pollutants to be focused on in this study are sulfate and SO_2 concentrations.

The estimation of local pollution is based on results generated by the UR-BAT model which imports sulfur emission data generated by RAINS-Asia and converts sulfur emissions into two forms of local air pollution: sulfate and SO_2 concentrations. Sulfate is both emitted directly–assumed at 5% of the total sulfur emitted–and formed from the chemical conversion of SO_2. Ambient concentrations of sulfate and SO_2 are calculated based on the distribution of sulfur mass in the surface and elevated layers.

Table 3 presents the sulfur emission and ambient concentration estimates. SO_2 emissions from China are projected to almost triple in the base case (from about 22 million tons in 1990 to over 60 million tons in 2020), and double in the efficiency case (to over 44 million tons in 2020). Local air pollution is assessed by focusing at two regions, Chongqing and Shanghai, both are large urban centers with high population density. Estimates of ambient concentrations were generated by the UR-BAT model which used the results of sulfur emissions for these two regions to develop isopleths of sulfate and SO_2 concentrations at fine resolution.

Measured against the threshold value of 60 $\mu g/m^3$ per year for SO_2 following the guidelines of the World Health Organization, both Chongqing and Shanghai are found to have SO_2 concentrations far exceed the standard set by the World Health Organization. By year 2020, population exposure to unsafe SO_2 concentrations is projected to reach over 16 million in Chongqing and nearly 14 million in Shanghai.

No guidelines have been set specifically for ambient sulfate concentrations. But evidence from a study by Dockery et al. (1993) shows that even for PM_{10}, particulates larger than sulfates ($PM_{2.5}$), there is a significant relationship between mortality rates and PM_{10} exposure, with a demonstrated increase in mortality of 0.8% for each 10 $\mu g/m^3$ annual increase in PM_{10} exposure. If to use 10 $\mu g/m^3$ as a measure for $PM_{2.5}$ exposure, both Chongqing and Shanghai were found to have far exceeded the 10 $\mu g/m^3$ value as shown in Table 3.

How accurate are the estimates presented in Table 3? Figure 2 compares the 1990 data of monitored annual average SO_2 concentrations at four sites in Shanghai with the predicted 1990 concentrations (Streets et al. 1998). Agreement is quite good for three of the monitoring locations, except for one site (the center of Shanghai) which the predicted value is much higher. The monitored data are sourced from the Global Environment Monitoring System (GEMS). The World Health Organization and the United Nations Environment Program (1992) caution that "the GEMS network is probably optimistic" as SO_2 concentrations monitored by the local Shanghai Environmental Protection Bureau appeared to be higher than the GEMS values at similar locations. The fact that the GEMS monitor values may be low makes the predicted values reasonable.

Streets et al. (1998) also report that less than 10% of the calculated SO_2 concentrations in the center of Shanghai were imported pollution from outside. Of the reminder, all industrial sources accounted for about 45% of the concentrations, with lesser contributions from the power sector (22%), residential sector (21%), large industry (11%) and transport (less than 1%). This clearly identifies the small, dispersed industrial sources as the greatest contributors to local pollution.

Extensive clinical, epidemiological and toxicological studies provide ample evidence of the relationships between exposure to ambient concentrations and human mortality and morbidity. Correlation between concentrations of different air pollutants, however, makes it hard to separate the effect of any single pollutant species on health. Data complied from the Chinese epidemiological studies and official statistics suggest that, of the many air pollutants, particulate matter and sulfate dioxide cause the most extensive health damage. Both are found to be associated with daily mortality and morbidity. Table 4 presents the dose-response coefficients estimated for Beijing, Chongqing and Shenyang.

- Beijing

  Xu and Gao et al. (1994) found that a doubling in SO_2 concentrations measured in $\ln(SO_2)$ was associated with increased risks of 29% in chronic obstructive pulmonary disease (COPD), 11% in cardiovascular disease (CVD), and 11% in total mortality. A doubling in total suspended particulates measured in ln(TSP) was associated with a 38% increase in COPD risk. In another study on morbidity effects, Xu, Li and Huang (1995) reported that a 100 $\mu g/m^3$ increase in TSP was associated with a 1.1% and 0.6% increase in hospital outpatients and emergency room visits in Beijing, respectively. Besides acute health effects, a third study done in Beijing on chronic health effects indicated that a 100 $\mu g/m^3$ increase in TSP concentrations was associated with an increased risk of bronchitis at an odds ratio of 1.9 (Xu and Wang 1993).

- Chongqing

  Stronger effects of SO_2 concentrations on daily mortality were found in the Chongqing study (World Bank 1996). A 100 $\mu g/m^3$ increase in SO_2 concentrations was associated with increased risks of 15%, 35% and 18% for COPD, CVD, and total mortality, respectively. Efforts to measure the health effect of $PM_{2.5}$ instead of TSP were made for the first time in China in the Chongqing study. However, no statistically significant association was identified between daily mortality and $PM_{2.5}$. The correlation between SO_2 and $PM_{2.5}$ at five sampling sites in Chongqing was found to be between 0.52 and 0.73 (Xu and Johnson 1997, p. 59-61).

- Shenyang

  Xu and Yu et al. (1997) reported that a 100 $\mu g/m^3$ increase in SO_2 and TSP concentrations were estimated to increase all cause mortality by 2.4% and 1.7%, respectively. Cause specific analysis showed that the association of SO_2 with COPD was significant (7.4%), but not statistically significant for CVD (1.8%), while the association of TSP with CVD was significant (2.1%), but not statistically significant for COPD (2.6%).

The magnitude of the health effects shown in Table 4 is not identical to that found in the western countries. The dose-response mechanisms are not yet fully understood. A common finding in the studies conducted in the three Chinese cities was that a statistically significant relationship between health outcomes and both SO_2 and particulates was observed. Whereas in the west the association between SO_2 and daily mortality was found to be weak or even disappeared if to consider simultaneously with particulates. The differences in the dose-response relationships reported for China relative to those in the west may reflect China's unique conditions in pollution sources and patterns. The ground level air pollution in China is primarily caused by fossil fuel (particularly coal) combustion emissions, with a strong seasonal trend of intense exposure to air pollution in the winter, whereas in industrialized countries automobiles fleet is the main culprit of ambient air pollution.

The interpretation here is that sulfur, generated primarily by coal combustion in China, is a good proxy for fine particulates, and that a large portion of the fine particulates are formed from sulfur in the atmosphere as sulfates. Thus, fine particulate ($PM_{2.5}$) is used as the key measure for dose-response estimation and economic valuation of the health impacts of air pollution in China. Following the World Bank (1997), the dose-response functions estimated by Chinese studies are used to assess the relationships between ambient air pollution and mortality and morbidity endpoints. International sources are used to supplement the Chinese studies for those endpoints which are not measured directly in China. The dose-response coefficients used in this study are listed Table 5. The estimated health impacts based on the dose-response functions show that about 180 thousand deaths would have been avoided, nearly 5.5 billion cases of acute and chronic morbidity would have been averted, and over 4.6 million person-years would have been saved, all in one year, if China had met its air pollution standards in 1995.

The economic valuation of the environmental damage to human health is based on a combination of willingness to pay and opportunity cost approaches. The willingness to pay approach is used to estimate the amount individuals are willing to pay for the benefit associated with lives saved as a result of air quality improvement.

A contingent valuation survey was conducted in Chongqing in 1998 to reveal people's willingness to pay to reduce the risk of death due to air pollution, with a sample size of 500. Both direct and indirect questionnaire methods were used in the survey. The results of the survey show that the willingness to pay for preventing a death from air pollution was about $26,000. But the use of the risk-dollar trade-off method similar to that designed by Viscusi et al. (1991) revealed a significantly higher implied willingness to pay at over $100,000. Taking into account of internationally comparable estimates of premature mortality costs, a median value of $60,000 for a statistical life in China is used in this study. This yields a total value of $10.8 billion for the lives lost due to air pollution. For the morbidity endpoints, the opportunity costs of illness approach is adopted to reveal the benefits of reduced medical care expenses and time spent in hospital visits, and regained productivity from bed confinement and work day losses of family members who take care of the sick persons. The costs associated with morbidity due to air pollution add up to more than $22 billion. Total damages (including both mortality and morbidity costs) associated with China's urban air pollution are estimated to be close to $33 billion.

The estimates presented in Table 5 are likely to be conservative, as indoor air pollution was not considered. Florig (1997) highlights China's indoor air pollution problem. Most industrial and residential boilers in China have adopted some basic particulate controls. But there are no particulate controls on China's hundreds of millions of household stoves. Also, our estimation focused on urban areas whereas the growth of township and village enterprises and rural population contribute to increased fuel burning and outdoor air pollution in rural areas. Pollution transportation by wind can be also a problem depending on the topography and weather conditions. The Chongqing $PM_{2.5}$ monitoring data (Xu and Johnson 1997, p. 47) show that, areas 150 miles away from Chongqing were found to have surprisingly high levels of $PM_{2.5}$ concentrations.

Regional Acid Deposition

At the regional level, the key to estimating the costs associated with acid deposition caused by SO_2 emissions is to resolve the uncertainty about the sources and effects of acid pollutants. The characteristics of sulfur emissions vary from country to country, the spatial patterns of acid deposition are volatile, and ecosystems are not uniformly assimilative of acid compounds. As a result, the costs and opportunities for acid rain control vary between countries. Evaluation of acid rain damage depends critically on how well we are informed of the following relationships:

$$D_i = \quad_j T_{ij} E_j , \qquad\qquad i = 1,2,...n; j = 1,2,...n$$

(1)

where D_i and E_j are vectors of acid deposition and sulfur dioxide emissions in countries *i* and *j*, respectively. T_{ij} is a transfer matrix which determines the proportion of net emissions from country *j* which are deposited in country *i*.

The amount of emissions, $E_{i,}$ depends on energy consumption, sulfur content and the technology for the removal of sulfur from emissions. It is possible to reduce emissions of sulfur, but at a cost. The total cost, C_i , is given by

$$C_i = C_i(E_i) + C_i(D_i), \qquad\qquad i = 1,2,...n$$

(2)

where $C_i(E_i)$ is the abatement cost function which is decreasing in E_i, and $C_i(D_i)$ is the damage cost function. Net cost function is derived by subtracting the damage cost function from the control cost function.

The RAINS-Asia model incorporates sufficient atmospheric and technical details which allow the model to generate estimates for the variables outlined in equations (1) and (2). Specifically, the RAINS-Asia model is used to generate four sets of results: estimates of future energy consumption under alternative socioeconomic and technical assumptions, sulfur emissions, the deposition of acid across countries and levels of excess acid deposition causing damages to the ecosystems, and the costs of emission control. RAINS-Asia projections of future energy consumption paths and sulfur emissions are reported earlier in Table 2 and Table 3, respectively. Acid deposition resulting from the baseline efficiency case is shown by both acid deposition originated from China (Figure 3) and the excess acid deposition above 25 percentile critical load in Asia (Figure 4).

As clearly mapped out in Figure 3 and Figure 4, if no action is taken to address the acid rain problem, acid pollutants emitted from China will fall not only in China but also in large areas of China's neighboring countries. This is because gaseous sulfur emissions can stay in the atmosphere for several days and travel hundreds of miles before falling back to the earth's surface. Acid deposition (in the forms of rain, snow, fog, dew, particles, or aerosol gases) is associated with many types of damages. These include crop and forest losses, soil and water acidification, corrosion to materials, and public health hazards. Chief among the acid deposition caused damages are the negative effects on agricultural crops and forests which occur directly through high ambient concentrations of sulfur dioxide and nitrogen oxides and indirectly through the acidification of soils. Soil acidification is likely to have long term impact on ecosystems and is believed to be the cause of the extensive dieback of forests in Central Europe over the past several decades.

The damages of acid deposition are assessed using the critical loads of acid deposition for a variety of ecosystems incorporated in RAINS-Asia. Following the recommendation that a critical load lower than 25 percentile entails uncertainty for the actual percentage of protection (Amann and Dhoondia 1994), excess acid deposition above 25 percentile (75% of protection for the ecosystem) in China is examined. As shown by Figure 4, sulfur deposition exceeds the critical loads in large areas sensitive to acidic precipitation in southwestern China, especially around Chongqing and where the excess deposition is estimated to be more than 11,000 $\mu g/m^2$ a year. Of the total land area in China, nearly 10% is estimated to have more than 1,000 $\mu g/m^2$ of excess sulfur deposition a year. The acid deposition estimates provided by the RAINS-Asia model as illustrated in Figure 4 are applied to an exponential dose-response function that captures the stylized relationship between acid deposition and crop/forest losses (World Bank 1997). The resulting estimates of China's crop and forest losses in the baseline efficiency case is estimated at about $4.2 billion a year (Table 6).

Figure 4 also shows that the ecosystems in the Korean peninsula and Japan are also subject to considerable risk imposed by acid rain disproportionate to their shares in sulfur emissions, particularly with considerable areas in South Korea show acid deposition at 500 $\mu g/m^2$ above critical loads.

Global Warming Impact

Based on fossil fuel consumption in the baseline energy pathway, carbon emissions from China can be estimated given the well-defined coefficients of carbon content in fossil fuels. The total carbon emissions from China's baseline fossil fuel consumption in 1990 and 2020 are estimated at 714 and 1607 million tons, respectively (Table 7). There is now consensus that the accumulation of carbon dioxide and other greenhouse gases in the atmosphere would give rise to global warming. The International Panel on Climate Change (IPCC 1996a) reports that the average temperature could increase 1~3.5 centigrade in about a century. While there are still uncertainties about the exact severity and timing, a rise of 1 centigrade would already be a major problem if the trend continues, and 3.5 centigrade would truly be a crisis.

China is a large country sensitive to climate change. Important agricultural areas such as the northern plain and northeast plain are in the "fragile climate region", with frequent extreme climate conditions and severe disasters. On average, there is one severe flood or drought every three years. The recent 1998 flood caused $20 billion in direct economic losses (Ji 1998). China also has 18,000 km of coastline. The southeast coastal region is a well-developed industrial and economic base with some of the densest population in China. These areas will be greatly affected by future climate warming and sea-level rise. Impact studies on climate change are at an early stage in China. Sponsored by the World Bank, the Chinese Research Academy of Environmental Sciences (CRAES) completed a preliminary assessment. The key findings of the CRAES (1994) study are highlighted here to provide an indication of the order of magnitude of the likely costs to China due to climate change.

Agricultural harvest in China relies basically on weather, and future climate warming will have a major impact on China's agriculture. While global warming will move China's warm temperature zones up north which will be favorable to China's agriculture in terms of less cold damage to crops and more planting areas, there will be more concentrated precipitation, greater evaporation, more frequent flood and draught conditions, and increased soil erosion and desertification. Also, insect pests and weeds are likely to increase due to temperature rise. Balancing the positive and negative factors, CRAES estimated that agriculture production in China is expected to drop by 5% or more under global warming scenarios.

For animal husbandry which depends critically on grass output, while increased CO_2 concentration will have a positive effect on grass growth, China's grass output may actually fall 10% or more when pests, diseases, and natural disasters are taken into account. Similarly, changes in rainfall, temperature and severe weather will damage aquatic ecological environment which will decrease fishery production. CRAES estimated that overall fishery production in China may decrease 5% or more by 2030 under conditions of climate warming.

At present, the total output value of agricultural production (including animal husbandry and fisheries) is about 17% of China's GDP. Using the present official GDP estimate of about $700 billion, a 5% reduction in agricultural production amounts to $6 billion in economic losses.

In the coastal areas of China, sea-level rise caused by global warming is expected to increase flooding, decrease port functions, and cause severe erosion and inundation. Based on IPCC's sea-level rise scenarios, it is estimated that sea-level rise in China may be in the range of 0.4~1 meter in the coastal areas by the year 2050. The inundated area can reach 92,000 km^2, close to 1% of China's total land. Saltwater can intrude inland as much as 30 km on the Yangtze River delta, causing water supply difficulties in cities like Shanghai. Also, increases of sea-level rise related disasters such as storm surges will cause heavy losses to life, property, and economic development in the coastal areas. A storm surge in 1992, for example, caused $1.2 billion economic losses (CRAES 1994, p. 89). To adapt to sea-level rise induced by global warming, more dikes and dams must be built. Chin has some 12,883 km of dikes at present along its 18,000 km coastline. It will cost at least $1.6 billion to raise the existing dikes, let along to build more to counter storm surges.

The impact of climate change on the ecosystem includes the direct and indirect impact on wetlands, vegetation, and biodiversity. As China's northeast regions become warmer, two of the world's greatest wetland areas, the Sanjiang plains marshes and Nuoergai marshes will decline. Sea-level rise will flood most of the existing sea shallows and change their ecological environment. As temperature increases, vegetation will shift from south to north by about 500 km by year 2050. Changes in forest vegetation will create difficulties for forest production. While the area of rainforests and monsoon rainforests will increase, highland plantation area in the southwest will decline. High quality wood production will decrease, and arid conditions in the north will increase the frequency and area of forest fire. The changes in both wetlands and vegetation will affect the habitats of wildlife and plant species, and threaten to reduce biodiversity.

The CRAES (1994) study, however, does not provide damage functions that allow us to link the level of China's greenhouse gas emissions to damage estimation. To compare the costs of China's air pollution at the global level to those at the local and regional level, a marginal cost approach as proposed by a number of studies (IPCC 1996b, p. 215) is used to quantify the economic losses associated with carbon emissions by China. A range of marginal costs per ton of carbon emissions is presented in Table 8. The marginal costs are estimated as the foregone future benefits that would otherwise be realized if one ton *less* of carbon is emitted in the present period. The marginal damages increase over time, reflecting the cumulative impact of temperature rise.

Although the models used to estimate the marginal damage of carbon emissions differ, the results are in broad agreement. Variations are largely accounted for by differences in discount rates. Cline (1992) finds higher shadow price for carbon using the discount rates of 0~2%, while Maddison (1994) and Nordhaus (1994) discount the future benefits at the rates of 4~5% and therefore find lower marginal cost of current carbon emissions.

The widely quoted average estimates of marginal cost for carbon emissions are those close to Fankhauser's (1994), rising from $20 to $28 per ton of carbon over the periods of 1991-2000 to 2021-30. Fankhauser used a probabilistic approach in which low and high discount rates were given different weights. Applying the marginal cost estimates by Fankhauser to China's carbon emissions, the estimated costs associated with China's carbon emissions in the baseline scenario would be about $14 and $45 billion at 1990 and 2020, respectively.

ASSESSMENT OF ABATEMENT MEASURES

Abatement Scenarios

Estimates of the damages caused by air pollution in China show that the greatest source of damage is from urban air pollution. Local health costs are estimated to be almost 8 times that of regional damages in crop and forest losses, and more than double the global damages associated with China's carbon emissions. Six scenarios are developed to tackle the air pollution problems in China. These scenarios are designed in a cost-effective manner according to the identified sources of damage. They extend from the abatement options in the *China 2020* study (World Bank 1997), and are refined in consultation with the State Environmental Protection Administration in China to make them as policy relevant as possible.

1) <u>Targeting small sources</u>. Focusing emissions control efforts on small sources may be more cost-effective than simply controlling/stabilizing total emissions. All but a small portion of fine particulate emissions is from small pollution sources. While power plants will account for most of the increase in SO_2 emissions, they do not

translate into a large increase in ambient SO_2 levels because of high stacks. On the other hand, higher SO_2 emissions from small sources (households and boilers) will result in high ambient levels and exposure to SO_2. So it would benefit more and cost less to target emissions from small sources than from large point sources.

2) Targeting critical areas. Differentiating nation-wide policy and specific measures for environmentally sensitive areas would be another important aspect of a cost-effective abatement strategy. Critical areas include urban cities with large population subject to high levels of pollution, and regions most sensitive to ecosystem damage such as the subtropical evergreen forests of southern China. A strategy with policies targeting critical areas would confer larger benefits than a uniform national policy as documented in *China 2020*.

3) Coal upgrading. This scenario includes two components. One is quality coal distribution which addresses the problem of regional concentration of sulfur pollution. Sulfur emissions are highest in the southern and southwest regions because of the high-sulfur coal produced and consumed there. The eastern region on the other hand has the highest energy consumption among all regions in China. Half of the coal requirement in the east needs to be imported from outside the region. A cost-effective option would be to transport low-sulfur coal to the southern, southwest and eastern regions to reduce sulfur emissions. Another component is clean coal technology (coal washing). Developing clean coal technology is part of China's long term energy strategy. As the world's largest coal producer and consumer, one of China's policy priorities would be to invest in coal washing, coal combustion, coal conversion, and post-combustion emission control technologies.

4) Intensive energy efficiency. Improving energy efficiency and conservation, especially increasing the efficiency of coal use would play a key role in reducing coal consumption thus coal-caused air pollution. This is extremely attractive as China will only gradually reduce its reliance on coal. Stronger efforts than that assumed in the baseline efficiency case are proposed in this scenario to accelerate energy efficiency improvement through policies such as market-based pricing to increase energy prices to reflect their true social costs.

5) Promoting natural gas and renewables (fuel substitution). Given China's heavy reliance on coal, increasing substitution of cleaner fuels - especially gas and district heating - for coal for household heating and cooking, and diversifying

energy supplies into non-coal sources such as renewables would reduce particulate and sulfur emissions as well as greenhouse gases.

6) Promoting hydropower. Hydropower has zero emissions, and China has rich hydro resources. Small scale hydropower projects have proved to be an important energy source for China's large population in rural and remote areas. One of the attractive abatement policies would be to further promote hydroelectric power.

Measures proposed in these scenarios can be classified into primary and secondary abatement. Primary abatement reduces emissions through changes in primary energy consumption by means of improving energy efficiency to reduce fuel consumption, and switching to low-sulfur or emission-free fuels. Secondary abatement does not change energy consumption, but removes pollutants from emissions either before burning (coal washing), or during burning (fluidized bed combustion), or after burning (flue gas desulfurization). Scenarios 1-3 fall into this category, while scenarios 4-6 belong to the primary abatement category. All abatement scenarios are implemented in the RAINS-Asia model.

Cost-Benefit Analysis

Cost evaluation in the RAINS-Asia model is restricted to the incremental costs related to secondary abatement measures. Therefore, the control costs for the scenarios of secondary abatement ("targeting small sources", "targeting critical areas" and "coal upgrading") are estimated using the cost parameters specified in the RAINS-Asia model. The control costs for the primary abatement scenarios are estimated outside the RAINS-Asia model based on cost parameters derived from a number of documented sources (Peng et al. 1998). These cost estimates are by no means definitive. They are only indicative of the order of magnitude of the likely abatement costs.

Table 9 reports the control costs and emission reductions in the abatement scenarios. The cost effectiveness of each scenario is measured by the costs per ton of emission reduction relative to the baseline scenario. It is clear that among the six abatement scenarios the "intensive energy efficiency" scenario is the most cost effective in reducing emissions of all pollutants, followed by the secondary abatement scenarios in the order of "targeting small sources", "targeting critical areas" and "coal upgrading", in reducing SO_2 and $PM_{2.5}$ emissions. The two scenarios that focused on changing fuel composition ("fuel substitution" toward natural gas and renewables, and "promoting hydro") appear to be far more expensive than other scenarios in reducing emissions.

Benefits associated with emission reductions are estimated in Table 10. Two factors are important in assessing abatement benefits. The first is population growth. Rapid urbanization in China implies that population growth in urban areas will be faster than that in rural areas. The increase in the size of urban population will have a large impact on the assessment of future benefits associated with air pollution abatement. Second, as air pollutants accumulate, their damaging impacts will increase over time. Both factors imply that there are large benefits to be realized if future emissions are to be reduced.

At the local level, it is assumed that a linear relationship exists between the growth of particulate and sulfur emissions and the increase of ambient concentrations and population exposure. The health benefits are estimated as the avoided costs associated with reductions in mortality and morbidity in year 2020 resulting from emission abatement. As expected, the local health benefits reported in column 1 of Table 10 are significant. Population growth and the expected increase in air pollution associated with future energy consumption dramatically increase the benefits of reducing future pollution.

In contrast, the benefits at the regional level are relatively small compared to that estimated for local health. The regional benefit estimates are derived by first extrapolating the damage estimate for 1990 to year 2020 at a constant $200 per ton of SO_2 based on the estimate of sulfur dioxide damage to farming and forestry (World Bank 1997, p. 27), and then calculating the avoided costs of crop/forest damages in line with the emission reductions relative to the baseline scenario.

At the global level, estimation of future benefits for carbon reduction takes into account that the impact on climate change increases disproportionately as the temperature increases over time. Fankhauser's estimates of (1994) marginal costs, which rise from $20 per ton of carbon in 1991-2000 to $27.8 per ton of carbon in 2021-2030, are used to estimate the benefits of CO_2 reductions in the abatement scenarios in 2020.

Given the estimates of costs and benefits, it is possible to make cost-benefit comparisons between the abatement scenarios. As the secondary abatement measures focus on reduction of $PM_{2.5}$ and SO_2 emission, the local and regional benefits associated with the secondary abatement scenarios are larger than those with the primary abatement scenarios. The benefits are particularly large in local health for the coal upgrading scenario ($54.3 billion, equivalent to 2.3% of China's GDP projected for year 2020 in the RAINS-Asia model). Given China's heavy reliance on coal, reducing sulfur emission from coal appears to deliver the highest benefits for public health and agriculture. This is also reflected in the results for the other two secondary abatement scenarios ("targeting small sources" and "targeting critical areas") where desulfurization technologies are adopted.

While the secondary abatement measures produce significant health and agricultural benefits, they also involve substantial abatement costs. This compares to the "intensive energy efficiency" scenario which brings about almost $41 billion benefits across local, regional and global levels, but at considerably less cost. The benefit/cost ratios show that, at the local level, the return rate is almost 200 per dollar for health investment in the "intensive energy efficiency" scenario, compared to the return rates between 6.6 and 8 per dollar in the secondary abatement scenarios. The advantage of the "intensive energy efficiency" scenario increases when the comparison extends to the regional and global level as shown by the overall benefit/cost ratio.

The cost-benefit comparisons highlight the importance of an integrated approach to air pollution control. The effectiveness of an abatement scenario needs to be evaluated in terms of emission reduction across different pollutants. Primary abatement, which reduces emissions through changes in primary energy consumption by improving energy efficiency or switching to cleaner or emission-free fuels, has the advantage of reducing all pollutants from fossil fuels, as demonstrated by the "intensive energy efficiency" scenario. The scope of fuel substitution into natural gas and renewable energy, however, appears to be limited. The benefit/cost ratios for the fuel substitution and hydropower scenarios are relatively low among all scenarios. This suggests that primary abatement by itself is unlikely to be sufficient to prevent further large increase in China's emissions. A more complete strategy will have to involve larger emission reductions, particularly through the adoption of abatement measures such as coal-washing, fluidized bed combustion, and sulfur scrubbing. These measures are categorized as secondary abatement. They do not affect primary energy consumption, but can achieve large emission reductions through technical improvement in removing sulfur and particulates.

Targeting small sources appears to be the most attractive option among the three secondary scenarios. Small, dispersed industrial and residential sources are the greatest contributors to ground-level ambient concentrations. Emissions from large sources such as power plants often do not translate into large increases in ambient concentrations due to high stacks. Given the large health impact at the local level, lowering ambient concentrations through reducing emissions from small sources therefore delivers substantial health benefits.

Economy-Wide Analysis

The cost/benefit comparisons of the abatement scenarios are based on *static* estimates of the control costs and environmental benefits. There are, however, important second round effects in adopting the identified abatement measures that will have indirect impacts on policy changes. Increasing energy efficiency or tackling small sources of emissions by increasing prices of fossil fuels to reflect their true social costs, for example, will induce behavior change in response to price increases. Consumers may respond by using less energy. The demand for energy in the economy will therefore decrease. This will help to lower the costs of abatement. In the case that energy is now used more efficiently, the benefits will include not only better air quality, but also increased productivity in energy end-user sectors. The total benefits will be larger than what was initially estimated. These effects are important in assessing alternative policy options from an economy-wide perspective.

Currently, the analysis in the RAINS-Asia model does not consider these second round effects. The demand for energy is fixed no matter how costly the abatement measures may be to energy users. Also, the indirect welfare impact resulting from abatement actions is not taken into account. Capturing these interactions between the abatement measures and their feedback from the economy requires an economy-wide analytical tool such as the GTAP model.

The GTAP model is a multi-sectoral and multi-regional applied general equilibrium model, which assumes perfectly competitive markets and constant returns to scale (Hertel 1997). GTAP has a number of features which make the model particularly suitable to address the second round effects associated with the abatement scenarios investigated in the RAINS-Asia model. First, GTAP uses a sophisticated demand system based on non-homothetic CDE (Constant Difference Elasticity) expenditure function to treat consumer preference explicitly, and allows for differences in both the price and income responsiveness of demand. Second, the

GTAP model incorporates a detailed welfare decomposition module to evaluate the welfare effects related to policy changes. Welfare in the model is measured in money metric term valued in US$. Third, GTAP has a highly disaggregated input-output database with detailed representations of energy sectors as well as extensive coverage of countries in the Asian region. To maximize the compatibility of country and sector representations between GTAP and RAINS-Asia, a tailored GTAP database featuring 15 regions and 10 sectors is generated from GTAP database version 4 (McDougall et al. 1998). [1]

A two-step approach is used to capture the economy-wide effects of the abatement measures. First, the same GDP and population growth assumptions adopted in the RAINS-Asia model are fed into the GTAP model to generate a standard forecasting simulation.[3] Then policy shocks are superimposed to the GTAP forecasting simulation to identify the changes. Two experiments are designed to examine the feedback from the economy:

- Coal tax. One way to put the RAINS-Asia abatement measures in action, for example, to accelerate energy efficiency improvement, is to use taxes to bring the prices of energy, particularly that of coal, to reflect its true social costs. A doubling of China's current coal price, suggested by the World Bank (1997) as the true cost of coal, is simulated to see the responding changes in energy consumption behavior.
- Energy input-augmenting technical change. This experiment is designed to capture the indirect welfare gains flowing from the RAINS-Asia "intensive energy efficiency" scenario by introducing a 5% increase in input-augmenting technical change for intermediate energy inputs across all sectors.

Table 11 presents the relative changes in energy demands in response to the coal tax policy shock. A doubling of the market price for coal reduces the demand for coal by 22% for private households and by 34% for the government sector. Demands for other energy products also decrease as a coal tax will increase the costs of other energy products which use coal as an input in their production. This is reflected in the larger decrease in the demand for gas and electricity than for oil and petroleum, as coal gasification and thermal power are important components in gas and electricity production in China.

More dramatic changes take place in energy imports and exports. As the domestic market price of coal doubles, demand for coal imports is predicted to increase significantly by almost 10 fold, while coal exports are diverted to supply to domestic market. This follows the Armington assumption on domestic and imported goods substitution specified in GTAP. A simpler specification is used at the intermediate input level using the Leontief assumption. Demands for all energy inputs are in fixed proportions relative to output. In response to a coal tax, the intermediate demands for all energy products decrease by about 3% in the energy intensive manufactures sector, as a higher coal price increases the production cost and reduces the output in this sector. In contrast, intermediate demands for energy inputs in other sectors increase slightly due to output growth. The Leontief assumption on intermediate input is admittedly strong for energy inputs in this study. A specification to allow interfuel substitution would capture the effects of relative price change between fuels (Peng and Martin 1994).

Table 12 reports the indirect welfare gains of a 5% energy input-augmenting technical change in addition to the environmental benefits of emission reduction associated with energy efficiency improvement. The simulation is illustrated in Figure 4 where the supply curve of an energy end-user sector, say energy intensive manufactures, is shifted downward as the marginal cost of production falls due to an energy input-augmenting technical change. The new equilibrium for the supply of and demand for energy intensive manufactures will shift from E to E*. The resulting welfare changes will include consumer's surplus increasing from APE to AP*E* and producer's surplus changing from PEB to P*E*B. Total welfare will increase by BEE*. This welfare gain in the energy intensive manufactures sector incurs in addition to the environmental benefits.

An equivalent variation (EV) welfare measure is incorporated in the GTAP model to evaluate the welfare effects related to changes such as an energy efficiency improvement. The net welfare gains of a 5% input-augmenting technical change are estimated to be about $7.5 billion (Table 12). As would expect, the largest source of welfare gains is from more efficient use of intermediate energy inputs simulated through a technical change. The increase of the competitiveness of the economy induced by the technical change also brings a positive impact on welfare through trade. Factor productivity also increases, as factors are shifted to relatively higher social marginal value usage. The positive impact of marginal utility of income reflects an increase of household income, and a greater proportion of household income spent on goods that increase household utility.

The negative allocative efficiency effect of over $3 billion losses in welfare is caused by the pre-existing trade protections in the economy. Perusal of the GTAP database reveals that there are significant export subsidies for energy intensive manufactures and coal, and considerable import tariffs for manufactures in general in the Chinese data. As the allocative efficiency effect in GTAP is measured by multiplying each tax (subsidy) with the relevant quantity change, the increase of exports and decrease of imports resulting from the technical change would give negative estimates for the allocative efficiency effects associated with exports and imports, due to the opposing signs in each pair of quantity change and tax/subsidy distortion. In the absence of other pre-existing distortions in the database, the trade transactions are solely responsible for the negative allocative efficiency effect.

The GTAP simulations capture important feedback from the economy in response to two abatement measures included in the RAINS-Asia scenarios. Consumer behavior does change once the externality of environmental pollution is incorporated in prices of fossil fuels to reflect their true social costs, and the indirect welfare gains from efficiency improvement in addition to the environmental benefits are substantial. To the extent that the benefits associated with the simulated abatement measures will be larger and the costs will be smaller than those estimated by RAINS-Asia, the insights provided by GTAP reinforce the policy message that there are significant net benefits in reducing air pollution, especially by improving energy efficiency.

CONCLUSIONS AND POLICY IMPLICATIONS

A clear conclusion of the analysis in this paper is that the largest source of damage by air pollution in China is human mortality and morbidity associated with ambient concentrations of fine particulates and sulfur dioxide. Adoption of abatement measures to improve urban air quality will have large benefits. The analysis also indicates that there is considerable synergy between local, regional and global abatement. The sources of emissions for the three pollutants - particulate, sulfur dioxide and carbon dioxide - are all by-products of fossil fuels, especially coal. Control policies that target the sources of these air pollutants will not only benefit the health of Chinese residents, but also reduce acid rain in East Asia and address the global warming problem.

Among the abatement scenarios examined, improving energy efficiency, especially increasing the efficiency of coal use appears to be the most cost effective. This is extremely attractive as coal is the main source of air pollution at the local, regional and global level, and China will only gradually reduce its reliance on coal. By increasing efficiency and reducing energy consumption, the intensive energy efficiency scenario targets at the source of the pollutants emitted by fossil fuels.

Economy-wide analysis indicates that improving energy efficiency also entails substantial indirect benefits flowing on to the rest of the economy through increased productivity in energy end-user sectors. In addition to technical improvement, increasing energy efficiency must emphasize removing distortions and breaking down market barriers. Welfare decomposition reveals that the benefits of an energy input-augmenting technical change can be significantly offset by policy distortions such as export subsidies or import tariffs.

Despite China's achievement in reducing energy intensity at a rate 5% annually in the past two decades, there appears to be scope for further improvement given the gap in energy intensity between China and other countries. China's current energy intensity more than doubles that of other Asian developing countries. Many projects that would save substantial energy and yield sound returns remain to be implemented, particularly in setting up information systems and strengthening energy management institutions. Internalizing the externalities of environmental pollution by increasing prices of fossil fuels to reflect their true social costs is critical in accelerating efficiency improvement.

Improving energy efficiency by itself will be insufficient to prevent further large increase in China's emissions. A more complete strategy will have to involve larger emission reductions, particularly through targeting small emission sources and critical areas. Adopting abatement measures such as coal-washing, fluidized bed combustion, and sulfur scrubbing are effective in supplementing energy efficiency improvement to reduce emissions. Also, transporting higher quality coal from the surplus producing northern region to the southwest and eastern coastal regions where local coal is high in sulfur can be an attractive option. The challenge is to transport coal over long distances, which adds to the already great pressures on the transport system. Reforming the transport sector to use market to determine the prices of transport services, and investing in transport infrastructure are integral elements of China's long term energy strategy.

The scope of fuel substitution into natural gas and renewable energy (including hydroelectric power) appears to be limited, and the costs involved in building these energy infrastructures are high. The key challenge will be to develop a sound government regulatory framework to encourage greater private sector participation and attract investment from international lending organizations to mitigate the risks.

Notes

1. Among the 15 regions identified, ten are major Asian regions represented in the RAINS-Asia model (China, Hong Kong, Taiwan, Japan, South Korea, Thailand, Indonesia, Malaysia, Philippines, and South Asia), plus North America, West Europe, Latin America, Africa and the rest of the world. Similarly, the sector aggregations broadly match those in the RAINS-Asia model, with five energy sectors (coal, oil, gas, petroleum, and electricity) and five energy end-user sectors (agriculture, energy intensive manufactures, less energy intensive manufactures, transport, and services). Details of GTAP analysis are reported in Peng et al. (1998).

2. In addition to GDP and population assumptions, the GTAP model requires assumptions for the growth of factor endowments (capital, labor and land) in order to generate forecasts. Capital and labor projections by Hertel et al. (1996) are extrapolated and adopted to form the factor endowment growth assumptions (agricultural land is assumed to be constant over time). Details of the forecast simulation are documented in Peng et al. (1998, p. 28).

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Figure 1. Analytical framework of integrated air pollution abatement in China

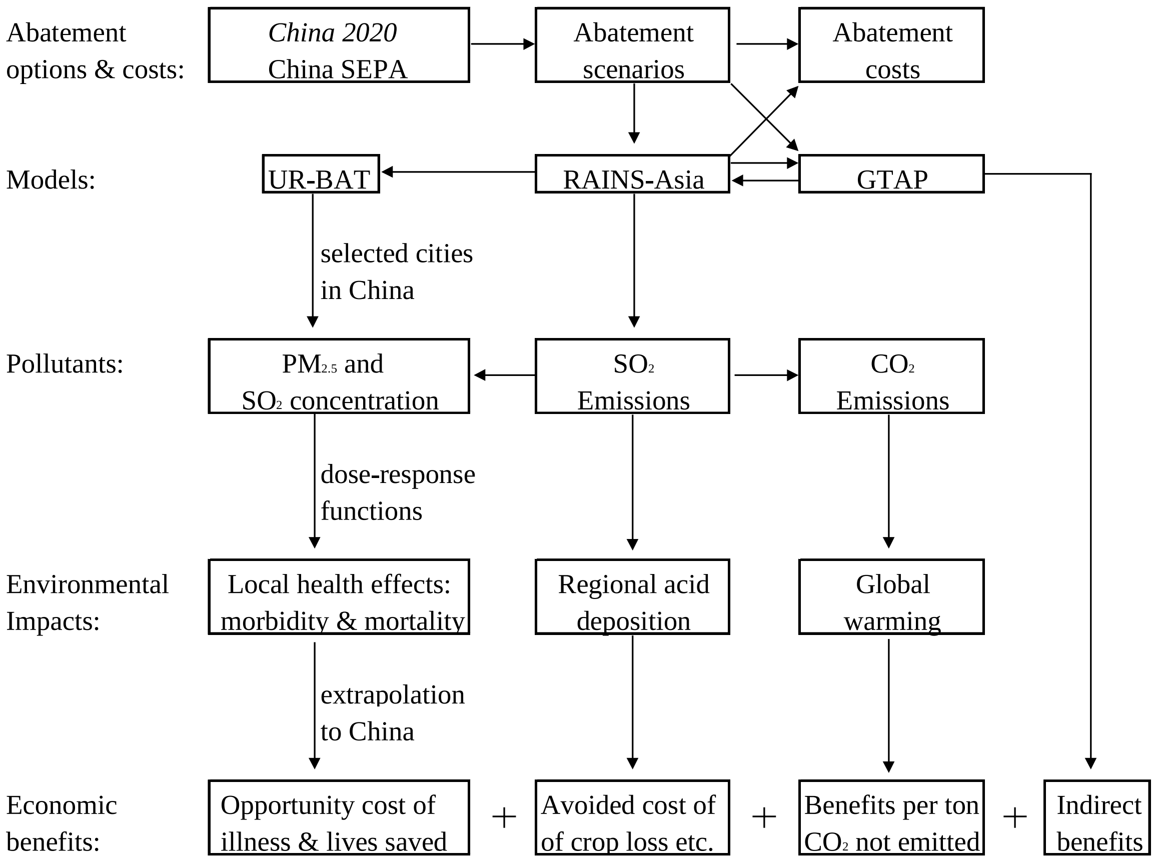


Figure 2. Observed and predicted average SO_2 concentrations, Shanghai 1990

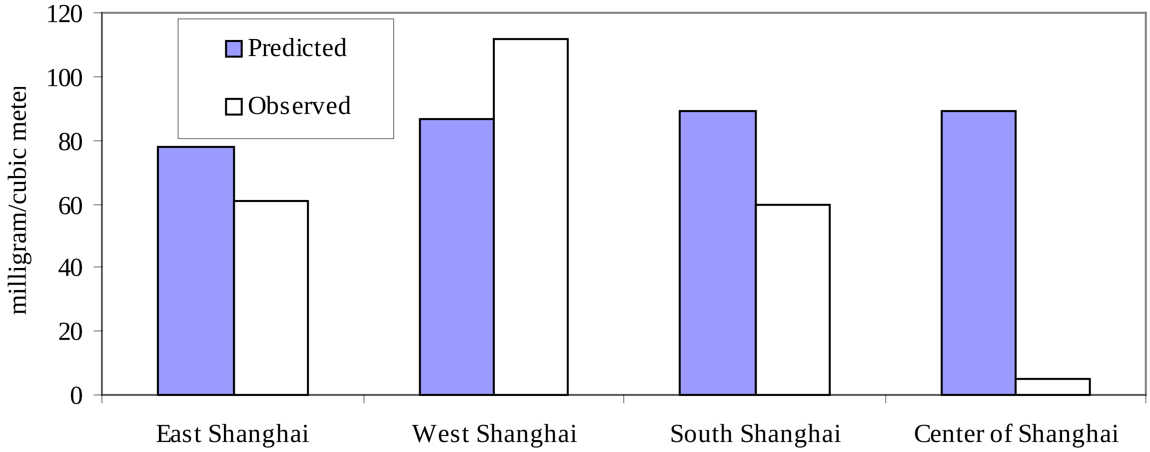
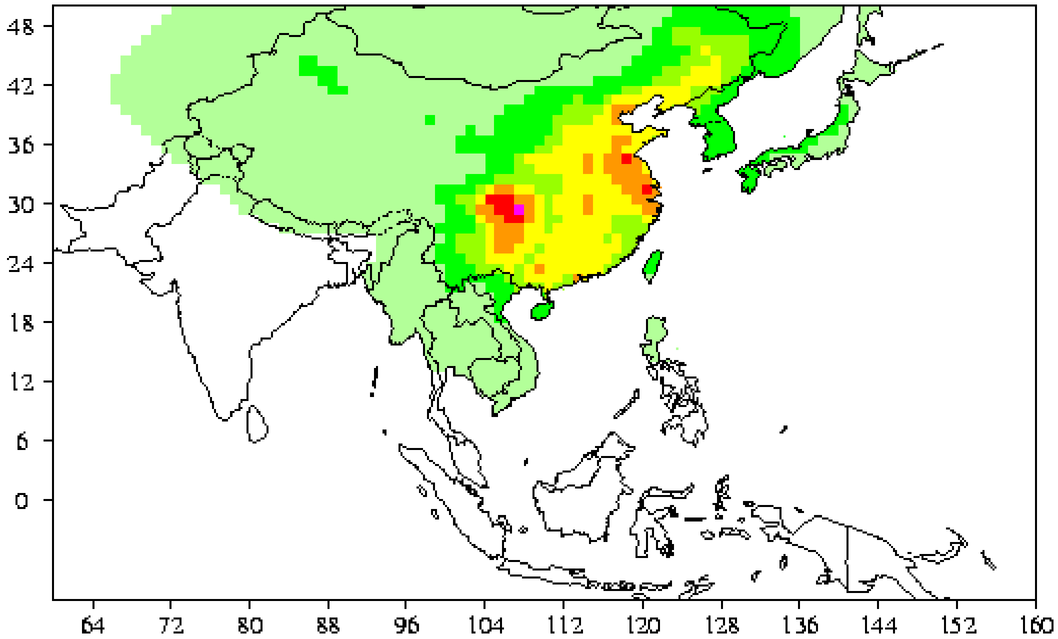


Figure 3. Acid deposition originated from China, baseline efficiency case, 1995



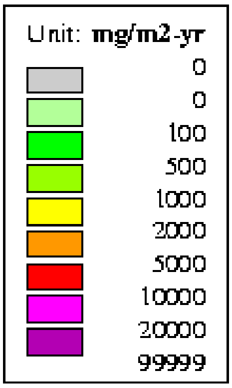


Figure 4. Excess acid deposition above 25 percentile critical load in Asia

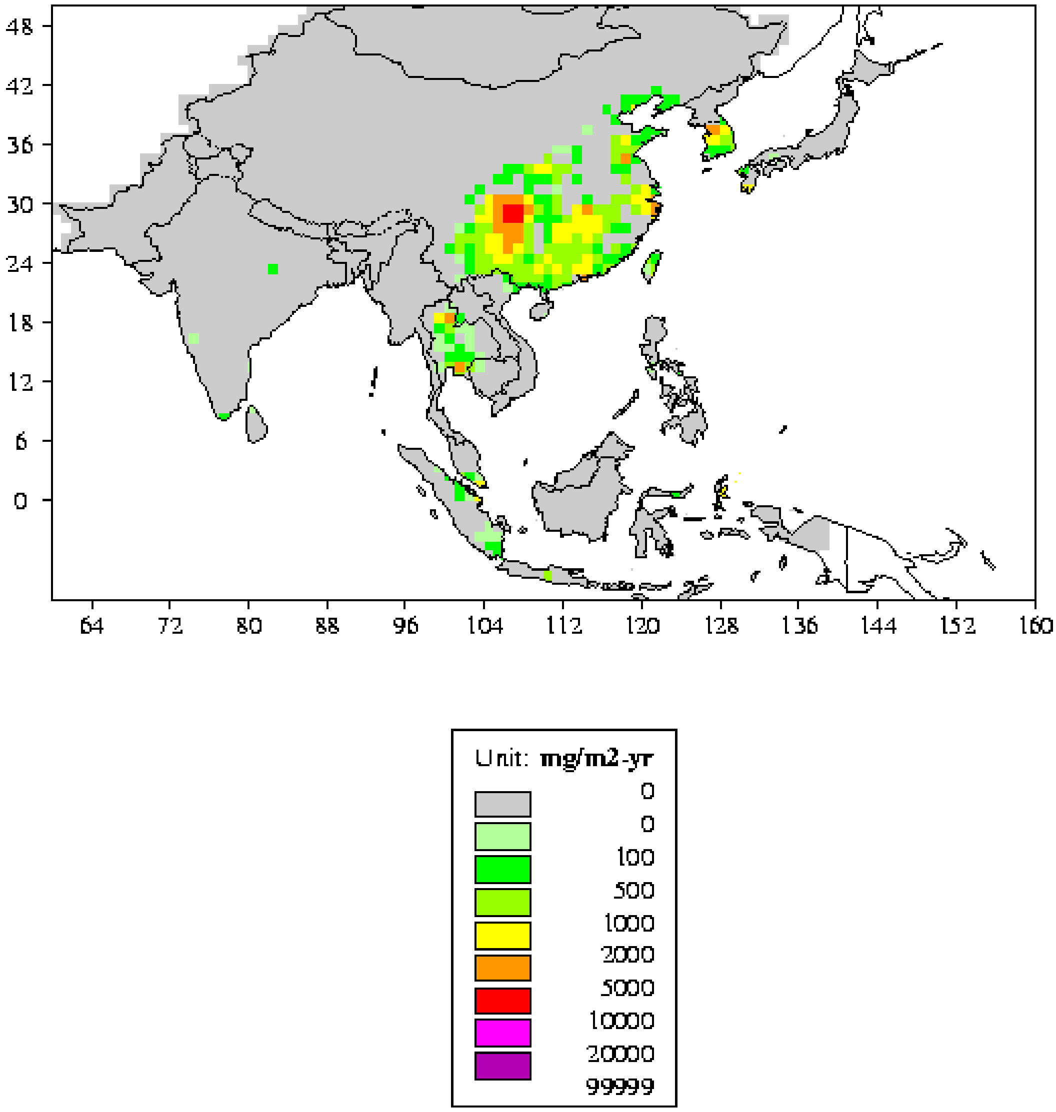


Figure 5. Welfare change associated with energy efficiency improvement

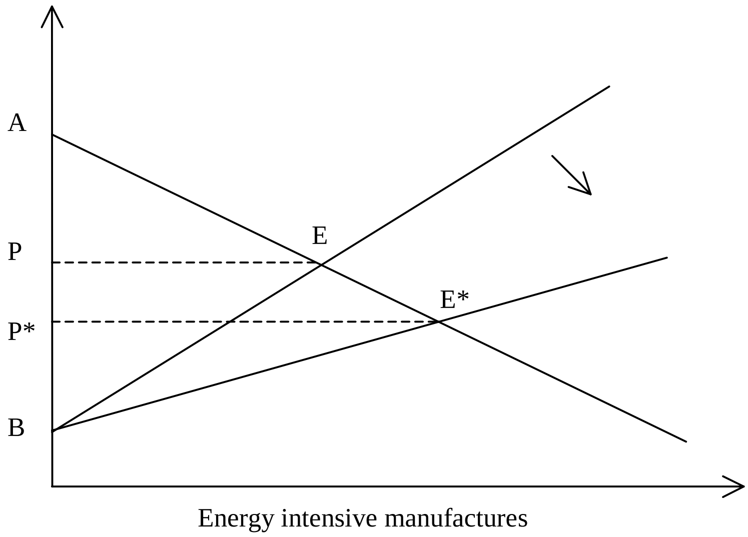


Table 1. Economic growth and energy consumption in a global context, 1994

	Energy use per capita (toe)	GDP per capita ($)	Energy intensity (toe/$1000)
China	0.65	440	1.48
South Asia	0.22	330	0.69
Bangladesh	0.07	220	0.29
India	0.24	320	0.76
Pakistan	0.26	410	0.62
Sri Lanka	0.11	650	0.17
East Asia developing	0.53	1350	0.39
Philippines	0.36	960	0.38
Indonesia	0.39	920	0.43
Thailand	0.77	2470	0.31
Malaysia	1.71	3530	0.48
East Asia newly industrialized	2.93	11010	0.27
Hong Kong	2.28	21980	0.10
Taiwan	2.44	11290	0.22
South Korea	3.00	8560	0.35
Singapore	6.56	22980	0.29
Asia Pacific industrialized	6.61	27980	0.24
Japan	3.83	36730	0.10
Australia	5.17	18440	0.28
Canada	7.80	18720	0.42
USA	7.91	25470	0.31
World	1.43	4500	0.32

Note: Regional totals refer to the sum of identified economies.
Source: Peng (1996, p. 12).

Table 2. Socioeconomic assumptions and energy pathways for China, 1990 to 2020

	1990-2000	2000-2010	2010-2020
Socioeconomic assumptions			
Annual growth in real GDP (%)	8.8	6.4	5.5
Annual growth in population (%)	1.2	0.9	0.6
Energy intensities			
Base case			
Energy per capita (GJ)	38.2	50.6	65.8
Energy per $1000 GDP (GJ)	67.6	52.6	42.5
Efficiency case			
Energy per capita (GJ)	32.8	40.7	49.1
Energy per $1000 GDP (GJ)	58.1	42.3	31.7
Energy pathways (annual energy growth %)			
Base case	5.3	3.8	3.2
Efficiency case	3.7	3.1	2.5

Source: Foell et al. (1995).

Table 3. Sulfur emissions and ambient concentrations in China, 1990 to 2020

	1990	2000	2010	2020
SO_2 emissions (kt)				
All China				
Base case	21908	34327	47840	60687
Efficiency case	21908	29523	38273	44676
Chongqing area				
Base case	974	1553	2135	2584
Efficiency case	974	1348	1705	1952
Shanghai area				
Base case	507	753	1145	1484
Efficiency case	507	621	905	1031
Local ambient concentrations, base case (population exposure, 10^6)				
Chongqing area				
SO_2 concentrations above 60 $\mu g/m^3$	5.1	12.3	15.1	16.4
Sulfate concentrations above 10 $\mu g/m^3$	14.3	17.1	17.4	17.5
Shanghai area				
SO_2 concentrations above 60 $\mu g/m^3$	0.65	3.46	9.75	13.72
Sulfate concentrations above 10 $\mu g/m^3$	0.65	2.54	8.29	14.08

Sources: SO_2 emission estimates are generated from RAINS-Asia. Ambient concentration estimates for Chongqing and Shanghai are based on Adhikary (1998) and Streets et al. (1998), respectively.

Table 4. Dose-response relationships measured in major Chinese cities

Doses		Responses	
	COPD	CVD	Total mortality
Beijing			
Doubling in SO_2	29%	11%	11%
Doubling in TSP*	38%	1.1% (HO)	0.6% (ERV)
Chongqing			
100 $\mu g/m^3$ increase in SO_2	15%	35%	18%
100 $\mu g/m^3$ increase in TSP	n.s.	n.s.	n.s.
Shenyang			
100 $\mu g/m^3$ increase in SO2	7.4%	1.8% (n.s)	2.4%
100 $\mu g/m^3$ increase in TSP	2.6% (n.s.)	2.1%	1.7%

COPD: Chronic Obstructive Pulmonary Disease. CVD: Cardiovascular Disease.
HO: Hospital Outpatients. ERV: Emergence Room Visits. n.s.: not significant (statistically).
* Coefficients for HO and ERV were measured in response to a 100 $\mu g/m^3$ increase in TSP.
Sources: Xu and Gao et al. (1994). Xu, Li and Huang (1995). Xu and Yu et al. (1997).
Xu and Wang (1993). World Bank (1996).

Table 5. Health impact of local air pollution in China, $PM_{2.5}$, 1995

	Dose-response Coefficients*	Health Impacts[#]	Economic valuation $ per unit	Economic valuation m$ in total
Mortality				
Premature death	8	180000	60000	10800
Morbidity (1000 cases)				
Acute				
Respiratory hospital admissions	16	354	284	100
Emergency room visits	305	6928	23	159
Lower respiratory infection/child asthma	30	678	13	9
Asthma attacks	3387	76885	4	308
Respiratory symptoms	237662	5394935	0.6	3237
Chronic				
Chronic bronchitis	79	1798	8000	14386
Restricted activity days (years)	74675	4644191	2.32	3933
Total				32932

*Effects per 1 million people for every 1 $\mu g/m^3$ increase in ambient concentrations of $PM_{2.5}$.
[#] Applying dose-response coefficients to a population of 250 million with excess $PM_{2.5}$ exposure of 91 $\mu g/m^3$.

Table 6. Estimates of acid rain damage to farming and forestry, 1995

Provinces	Annual sulfur Deposition ($\mu g/m^2$)	Extend of damage to crops and forests (%)	Gross output of crops and forests ($ billion)	Total damage ($ billion)
Liaoning	1500	1.9	48.2	0.9
Beijing	1700	2.2	10.7	0.2
Tianjin	3100	5.2	10.2	0.5
Hebei	1500	2	92.6	1.9
Shanxi	1000	0.8	25.7	0.2
Shanghai	4600	8.5	9.3	0.8
Jiangsu	4200	7.6	119.9	9.1
Zhejiang	2500	4	63.3	2.5
Anhui	2100	3.2	80.7	2.6
Fujiang	1300	1.5	47.5	0.7
Jiangxi	1800	2.5	44.4	1.1
Shandong	2300	3.8	116.0	4.4
Henan	2000	3	107.6	3.2
Hubei	1800	2.5	76.2	1.9
Hunan	1500	1.9	74.0	1.4
Guangdong	1900	2.6	98.1	2.6
Guangxi	1600	2	49.6	1.0
Sichuan	2900	5	111.0	5.5
Guizhou	3000	5.1	28.5	1.5
Yunan	400	0	40.5	0.0
Shaanxi	1300	1.5	32.7	0.5
Total	44000		1286.8	42.6

Table 7. Estimates of carbon emissions in the baseline energy pathway for China*

Fuel type	Carbon emission Coefficients	Fuel consumption		Total carbon emissions	
		1990	2020	1990	2020
	(000' tons per PJ)	(PJ)	(PJ)	(million tons)	(million tons)
Coal	25.8	23342	48239	602	1245
Heavy fuel oil	21.1	2781	2998	59	63
Medium distillates (diesel oil, light fuel oil)	20.2	843	6148	22	124
Light fractions (gasoline, kerosene, naphtha)	18.9	11098	5395	16	102
Natural gas	15.3	949	4755	15	73
Total		29013	67535	714	1607

*Carbon coefficients are based on World Bank (1994, Table 2.3, p. 33), and fossil fuel consumption is generated by the RAINS-Asia model.

Table 8. Damage estimates of carbon emissions from China, at 1990 prices

Selected studies	Marginal cost ($ per ton of carbon)		Total damage ($ billion)*	
	1991-2000	2021-2030	1990	2020
Cline (1992)	5.8~124	11.8~221	4~89	19~355
Fankhauser (1994)	20.3	27.8	14	45
Maddison (1994)	5.9~6.1	14.7~15.2	4	24
Nordhaus (1994)	12	n.a.	9	n.a.
Peck and Teisberg (1992)	10~12	18~22	7~9	29~34

* Multiplying the marginal cost estimates for 1991-2000 and 2021-2030 by China's carbon emissions in 1990 (714 million tons) and 2020 (1607 million tons), respectively.

Table 9. Control costs and emission reduction in abatement scenarios, China, year 2020

	Control costs and emissions				Cost effectiveness*		
	Costs ($ billion)	$PM_{2.5}$ (million tons)	SO_2 (million tons)	CO_2 (million tons)	$PM_{2.5}$ ($/ton)	SO2 ($/ton)	CO_2 ($/ton)
Targeting small sources	5.4	1.6	31.7	5891	8318	416	n.a.
Targeting critical areas	4.7	1.7	34.8	5891	9491	475	n.a.
Coal upgrading	8.2	1.4	27.6	5891	9640	482	n.a.
Intensive energy efficiency	0.2	1.8	35.7	4930	359	18	0.17
Fuel substitution	7.7	2.1	41.2	5773	43884	2194	64.97
Promoting hydro	9.6	2.1	42.5	5705	89055	4453	51.48

* In terms of cost per ton of emission reduction relative to the baseline scenario where $PM_{2.5}$, SO_2 and CO_2 emissions are 2.2, 44.7 and 5891 million tons, respectively.

n.a. Not applicable. Cost effectiveness is not estimated for CO_2 emission reduction in the first three scenarios where secondary abatement measures are adopted. Secondary abatement measures do not change fossil fuel consumption relative to that in the baseline scenario. Neither do they reduce carbon emissions. Notice CO_2 emissions in these three scenarios are all estimated to be 5891 million tons, equivalent to 1607 million tons of carbon as estimated in for the baseline scenario (the molecular weight of CO_2 is 44 units, where C weighs 12 units, and O_2 weighs 16X2 units).

Table 10. Benefit/cost comparisons between abatement scenarios, China, year 2020

	Benefits ($ billion)				Benefit/cost ratios			
	Local	Regional	Global*	Overall	Local	Regional	Global	Overall
Targeting small sources	43.7	2.6	0.0	46.3	8.1	0.48	0.0	8.6
Targeting critical areas	34.4	2.0	0.0	36.4	7.4	0.42	0.0	7.8
Coal upgrading	54.3	3.4	0.0	57.7	6.6	0.41	0.0	7.0
Intensive energy efficiency	31.8	1.8	7.3	40.9	197.0	11.08	45.1	253.2
Fuel substitution	13.3	0.7	0.9	14.9	1.7	0.09	0.12	1.9
Promoting hydro	8.3	0.4	1.4	10.1	0.9	0.04	0.15	1.1

*Benefits at the global level associated with carbon emission reductions are not estimated for the first three scenarios where secondary abatement measures are adopted, and they do not change fossil fuel consumption relative to that in the baseline scenario.

Table 11. Changes in energy demand in response to a doubling of coal price in China (%)

	Coal	Oil	Gas	Petroleum	Electricity
Final demands					
Private households	-22.1	-3.5	-8.2	-3.4	-9.9
Government sector	-34.1	-6.9	-6.7	-4.9	-19.5
Imports	981.8	4.2	39.1	-3.5	114.2
Exports	-137.8	-7.4	-171.3	2.3	159.2
Trade balance	-285.9	-4.9	45.3	5.2	113.3
Intermediate demands					
Agriculture	1.6	1.6	1.6	1.6	1.6
Energy intensive manufactures	-2.9	-2.9	-2.9	-2.9	-2.9
Less energy intensive manufactures	4.9	4.9	4.9	4.9	4.9
Transport	0.3	0.3	0.3	0.3	0.3
Services	0.6	0.6	0.6	0.6	0.6
Energy supply					
Domestic production	-11.1	-3.8	-20.9	-0.9	-4.4
Supply price	11.8	1.6	20.7	0.5	24.5
Market price	108.2	1.6	20.7	0.5	24.5

Source: GTAP simulation by the author.

Table 12. Welfare changes of a 5% energy input-augmenting technical change in China

	$ million	% in total EV
Equivalent Variation (EV)	7472	100
All technical change	4966	66
Terms of trade effect	3908	52
Factor productivity	1563	21
Marginal utility of income	227	3
Allocative efficiency effect	-3192	-43

Source: GTAP simulation by the author.

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