World Health Organization
Sustainable Development and Healthy Environments

WHO/SDE/PHE/99.1
Original: English
Dist.: Limited

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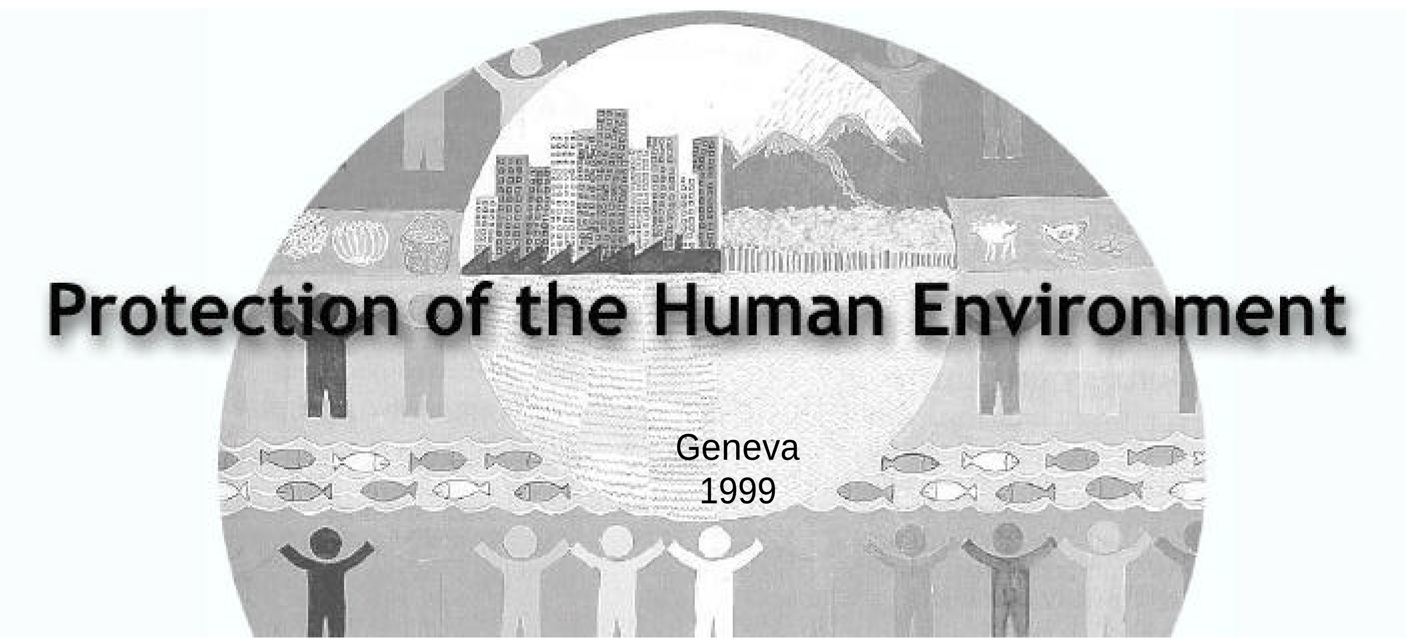

Geneva
1999

WHO/SDE/PHE/99.01
DISTR: LIMITED
ENGLISH ONLY

Near-term Health Benefits of Greenhouse Gas Reductions:

A Proposed Assessment Method and Application in Two Energy Sectors of China

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Geneva, March 1999

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PREFACE

This is a study of projected near-term health benefits associated with greenhouse gas (GHG) reductions resulting from changes in energy efficiency and structure of energy use in the power and household sectors of China. The work was commissioned by the former Office of Global and Integrated Environmental Health at WHO, in order to explore the scope for modelling in the assessment of such short-term health benefits. China was selected as an appropriate case study for this work, as it fulfilled most of the criteria required, including the fact that it is a large country, with data sets available on air pollution and health, and with information on projected trends in the consumption of fossil fuels.

The analysis in question represents a first step in the development of a creative approach to estimating health impacts due to particulate and SO2 emissions in the power and household sectors, and in providing an economic analysis of the relative merits of promoting various changes in fuel use in these sectors. It presents important work in the development of a methodological framework and has significant implications regarding GHG control policies and associated health benefits. The model assumptions made in the present study do not however necessarily reflect existing short-and long-term energy, climate and health obligations of any Party to the United Nations Framework Convention on Climate Change.

The paper develops several alternative scenarios, and makes the important point that there may be substantial health benefits of GHG control policies which are associated with concomitant reductions in health damaging air pollutants which will occur, the degree of health benefit depending markedly on the particular scenario of GHG reduction options chosen (for example the choice of energy technologies and sectors). In China, improving the efficiency of, or switching away from traditional use of coal and biomass, could result in substantial health benefits. Similar health benefits could be expected to accrue to other countries with high dependance on solid fuels in the household and power sectors.

As the authors recognise, China is a diverse country, with widely ranging (and changing) physical, social and economic conditions, including varying and distinctive patterns of energy use, a variety of geographic and meteorological conditions, a diversity of living conditions, lifestyles and socioeconomic situations. This poses limitations on the extent to which generalisations can be made for the country as a whole, and on the types of extrapolations which are possible, considering the data limitations and the relatively few studies conducted to date.

There were a number of constraints under which the study operated. These included limitations in air monitoring data bases, emissions inventories of indoor and outdoor fossil fuel sources of airborne particulates, gaps in population and health effects data. Assumptions were made regarding factors such as TSP emissions, meteorological factors, average daily cooking times and other aspects which influenced the air dispersion analyses and exposure assessment estimates. Such assumptions are highlighted as relevant throughout the text. As the authors also point out, the statistics are varied by year, source, location, methods and approach.
There are indeed many possibilities for future studies, which were outside the scope of the current work. These might attempt a disaggregated analysis, in which regional differences

could be examined in more detailed subsequent analyses. In particular, the rapidly growing urban areas might be assessed in more depth, as well as the rural areas, and the health implications of the accompanying expected shifts and trends in fuel use assessed. These are difficult to predict as the mix of fuels used is constantly shifting.

The choice of energy technologies is clearly of great importance. There is a need in this regard for a fuller accounting of the near-term health benefits of GHG reductions. Future studies might look at additional pollutants beyond SO2 and particulates, as well as other health effects, for example lung cancer, acute CO poisoning, arsenic and flourine exposure due to indoor coal combustion, impacts on media such as water, and waterborne diseases, and so on.

In addition, of great importance is that the analysis be extended to other sectors, for example agriculture, forestry and, in particular, the transport and industry sectors, for which predicted growth is large, with significant implications for the control of greenhouse gases in these sectors.

China is a country undergoing momentous change. Industry is in the course of reforming and renewing its technology, and the country has made important attempts to improve household energy efficiency and fuel use patterns. In many coastal areas for example, there has been a marked shift to natural gas and use of LPG. In order to provide better estimates of the health gains associated with GHG control options in the future, there is a need for better emissions data bases, and for more comprehensive and systematic monitoring systems. There is also a need for better information on the links between emissions, ambient concentrations, human exposures and health impacts.

It is hoped that this report will stimulate further studies which address also alternative scenario options associated with health benefits expected in other key sectors such as industry and transport. It is also hoped that this report will encourage countries in the formulation of appropriate programmes to improve the quality of local emission factors, data and models, and in the formulation of their national climate change programmes in line with the requirements of the Kyoto Protocol.

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EXECUTIVE SUMMARY

Human additions to global greenhouse-gas emissions are contributing to the risk that climate may change more rapidly than it ever has in human history. One principal set of adverse outcomes from such change seems likely to be impacts on human health. Thus, studies have shown that reduction in current greenhouse gas (GHG) emissions could lead to long-term health benefits in the form, for example, of reduced extent of malarial mosquitoes, fewer extreme climate events, and lower impacts on food production compared to what might occur if today's GHG emission trends continued. Each step of the causal chain from GHG emissions through global warming to health effects is not understood with certainty, however, leading many observers to view the overall connection with skepticism.

There is another and more certain health benefit from GHG reductions, however, in the concomitant reduction in health damaging air pollutants that will occur as well. To obtain an idea of the extent of this less speculative benefit, we estimate here the near-term human health benefits of GHG reductions resulting from changes in energy efficiency and structure of energy use. We examine the power and household sectors of China, both as a case study for the method and because China is such an important actor in global GHG scenarios.

The methodological framework followed in this report is shown in the Figure A. A GHG reduction target is chosen for China: 10% below business-as-usual (BAU) by 2010 and 15% below by 2020. Four scenarios are examined overall: BAU, supply-side energy efficiency improvement, least-cost per unit global-warming-reduction fuel substitution, and least-cost per unit human-air-pollution-exposure-reduction fuel substitution scenarios. This comparison allows us to examine the relative near-term health benefits achieved by different technological and policy approaches to meeting GHG reduction targets. By near-term, we refer to the reduction in acute and chronic health impacts from air pollution for the first two decades of the 21st century.

The next step is to conduct a set of technology assessments comparing the health damaging (HDP) emissions (particulates and sulfur dioxide, SO_2), GHG emissions (carbon dioxide, CO_2 , and methane, CH_4), and economic costs of each energy technology option on a per-unit-energy basis. Then, the GHG emissions are converted to overall Global Warming Potential (GWP), and HDP emissions to actual human inhaled doses. Thus, an ordering of technologies is constructed in two pathways: least-cost per unit GWP reduction and least-cost per unit dose reduction, both of which cumulatively reaches the GHG reduction target. As shown in Figure A, once the change in inhaled dose is determined, the resulting change in health risk can be calculated using exposure-response information from available epidemiological studies and other information. Finally, this report estimated the marginal net economic costs of GHG reduction when moving away from conventional coal use, which are the differences between the economic benefits from improved health and the incremental costs associated with changes in energy systems.

Results: Following this methodological framework, we find that GHG reductions resulting from changes in energy use are generally accompanied by substantial near-term human health benefits. The degree of health benefit varies greatly with the choice of energy technologies and sectors, however. Shifting from conventional coal-fired power plants to natural gas, for example, has larger health benefits than GWP reduction, while shifting from coal power to hydro results in the same percentage reduction in both HDP and GHG emissions. This variation in health benefits is even larger between sectors. Our conservative estimates show, for example, that the health benefits of one ton reduction in particulates emissions from household stoves are at least 40 times larger than those from coal-fired power plants.

The following table shows the estimated annual avoided premature deaths from the three alternative scenarios relative to the BAU case by 2020. Compared to the projected mortality of 14 million in China by 2020, the annual avoided death of some scenarios could reduce total mortality by as much as 4%. Similar comparisons for various measures of morbidity are presented in the main report.

Avoided Annual Premature Death of the Three Scenarios by 2020

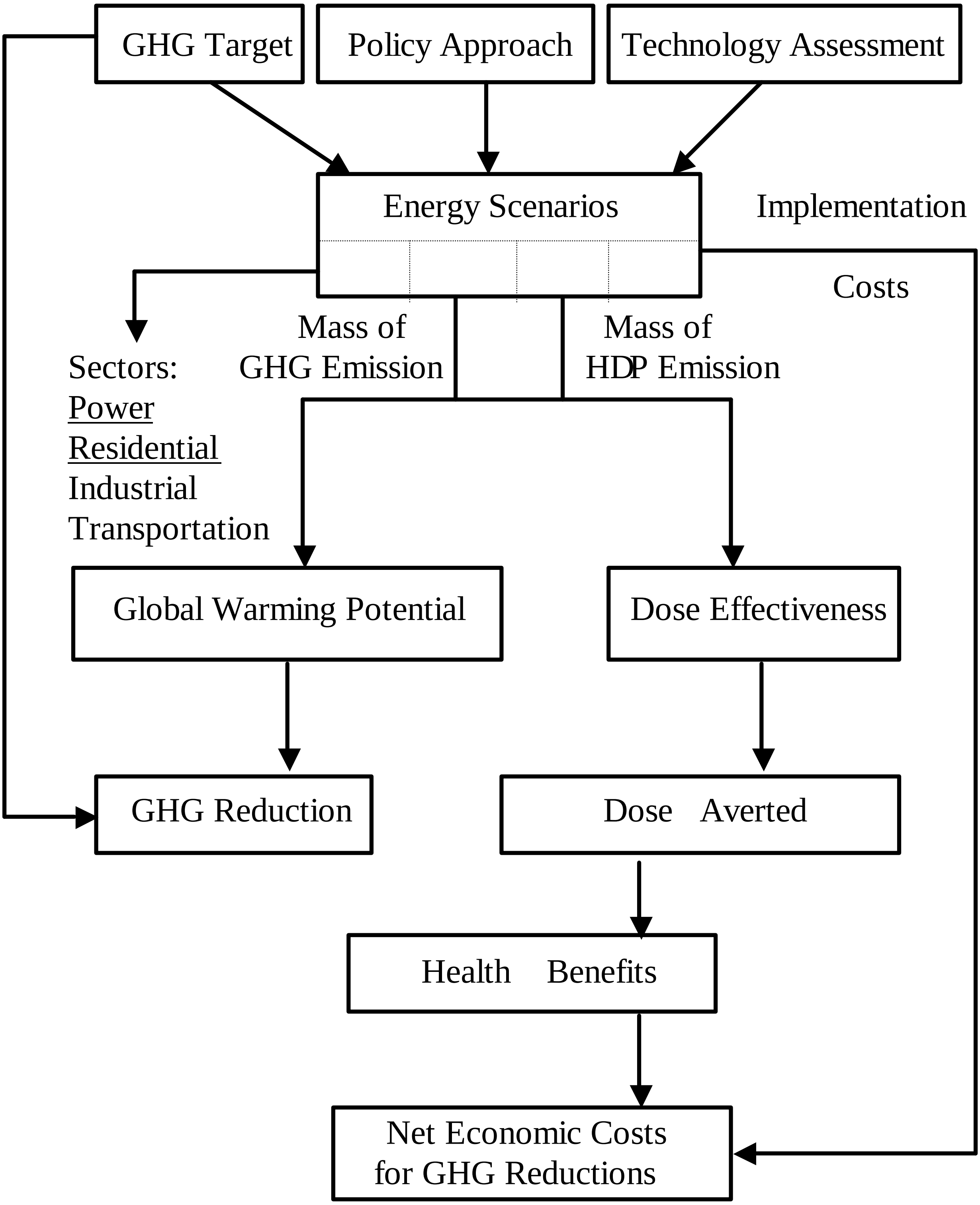
Scenario	Sector	Low	Central	High
Efficiency	Power	1,500	4,400	13,000
	Household	62,000	150,000	460,000
Substitution				
Least-cost GWP	Power	1,700	4,900	15,000
	Household	47,000	120,000	360,000
Least-cost dose	Power	1,800	5,200	16,000
	Household	70,000	180,000	530,000
Total mortality in China in 2020: 14,000,000				
Total population in China in 2020: 1,470,000,000				

This analysis shows that in terms of human health benefits choice of the energy technologies and sectors in which to conduct GHG reduction efforts is more important than choice of a particular target of GHG reduction.

Based on the health benefit estimates, we also evaluated the marginal economic benefits of such improved health impacts. Although GHG reduction itself is usually accompanied by increasing costs, when the economic benefits of improved health are counted, there is a net economic benefit for the those household energy options involving a shift away from traditional coal use. The economic benefits of improved health in the electric power sector, however, seem not to be large enough to offset the incremental economic costs associated with GHG reduction.

Conclusion: This study shows that the near-term health benefits from GHG reductions in China could be substantial and are highly dependent on the technologies and sectors chosen. In China, much of the health benefit occurs by improving the efficiency of or switching away from traditional use of coal and biomass. Thus, other countries with high dependence on solid fuels in the household and power sectors, India for example, could be expected to have a similar scale of benefits. Such near-term benefits provide the opportunity for a true "no-regrets" GHG reduction policy in which substantial advantages accrue even if the risk of human-induced climate change turns out to be less than many people now fear.

Figure A. Analysis Framework



1. INTRODUCTION

Human actions may be pushing the world's climate to change more rapidly than ever before in human history. Estimates from the Intergovernmental Panel on Climate Change (IPCC) are that the mean surface temperature of Earth could increase 1-3.5 ^{0}C over the next century due to increasing greenhouse gas (GHG) emissions resulting principally from human energy and food production (IPCC, 1996). In an effort to reduce the risk, 167 nations signed the United Nations Framework Convention on Climate Change in June, 1992 (United Nations, 1995). In December, 1997, 36 developed countries reached an agreement, which if ratified, will commit them to an average 5% reduction below 1990 levels (Bolin, 1998). Developing countries, however, have not yet agreed to specific targets or timetables for limiting their emissions (MacKenzie, 1997).

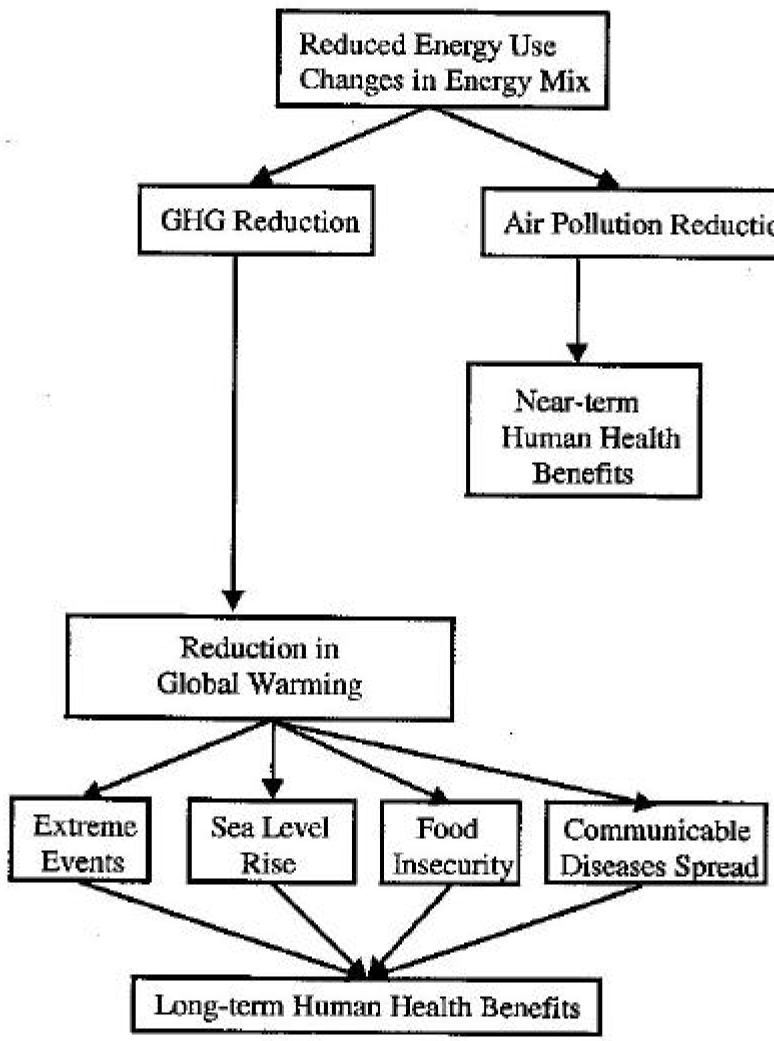
The most persuasive argument for spending resources to reduce current emissions is that the benefits in the form of reduced impacts of climate change will be greater. Among the important benefits, which also include avoiding ecosystem effects that could have significant indirect impacts on humanity, are avoiding or reducing the direct impacts on human health that might accompany climate change. As indicated on the left side of Figure 1, the principal routes through which such health impacts might occur seem to be shifts in the geographical extent of disease vectors such as malarial mosquitoes, an increase in extreme events including tropical cyclones and heat episodes, shifts in atmospheric composition toward more pollution, more costly food production, and increasing refugee populations from sea-level rise and other factors (McMichael et al., 1996).

The chain of causation from GHG emissions through global warming to direct health effects has several steps, however, each of which is not understood with certainty. As a result of the consequent overall uncertainty and long time delay, many observers are still unconvinced that the potential, but far-off, health benefits, justify large expenditures today on GHG reductions. Although the consensus scientific opinion as represented by the IPCC is that such ill-effects are likely if current GHG emissions trends continue, such skepticism still hinders international agreements to significantly alter current patterns of GHG emissions. This is particularly so in developing countries, which must contend today with many urgent problems related to human health and welfare.

One approach to this impasse is to promote "no-regrets" GHG reduction scenarios, which achieve a significant degree of near-term benefits as well as GHG reduction so that immediate action can be justified even if it later develops that the climate sensitivity to GHG additions is less that now thought (Repetto & Austin, 1997). Examples of such near-term benefits are the environmental and energy security advantages that would accrue through less dependence on fossil fuels and the human welfare benefits that could come about if an international GHG control regime were oriented toward assisting economic development and reducing vulnerability among poor populations (Hayes & Smith, 1994).

Also among the significant near-term benefits of GHG reductions are the human health benefits resulting from changes in the efficiency and structure of energy use that would be a large part of most GHG reduction scenarios. Although fuel cycles impact health in several ways, for example, through water pollution, the potential for large accidents, and occupational health and safety, probably the largest and most sensitive to change are those related to air pollutant emissions from processing and burning of fuels. The same combustion processes that produce GHG emissions such as carbon dioxide and methane also generate local and regional airborne health-damaging pollutants (HDP) such as particulates and sulfur oxides, as indicated by the right side of Figure 1. Thus, reduction in GHG can be expected to achieve HDP reduction as well, thus potentially bringing near-term health improvements. In this report, by “near-term,” we mean health benefits that manifest themselves by 2020.

Figure 1. Global Warming and Human Health



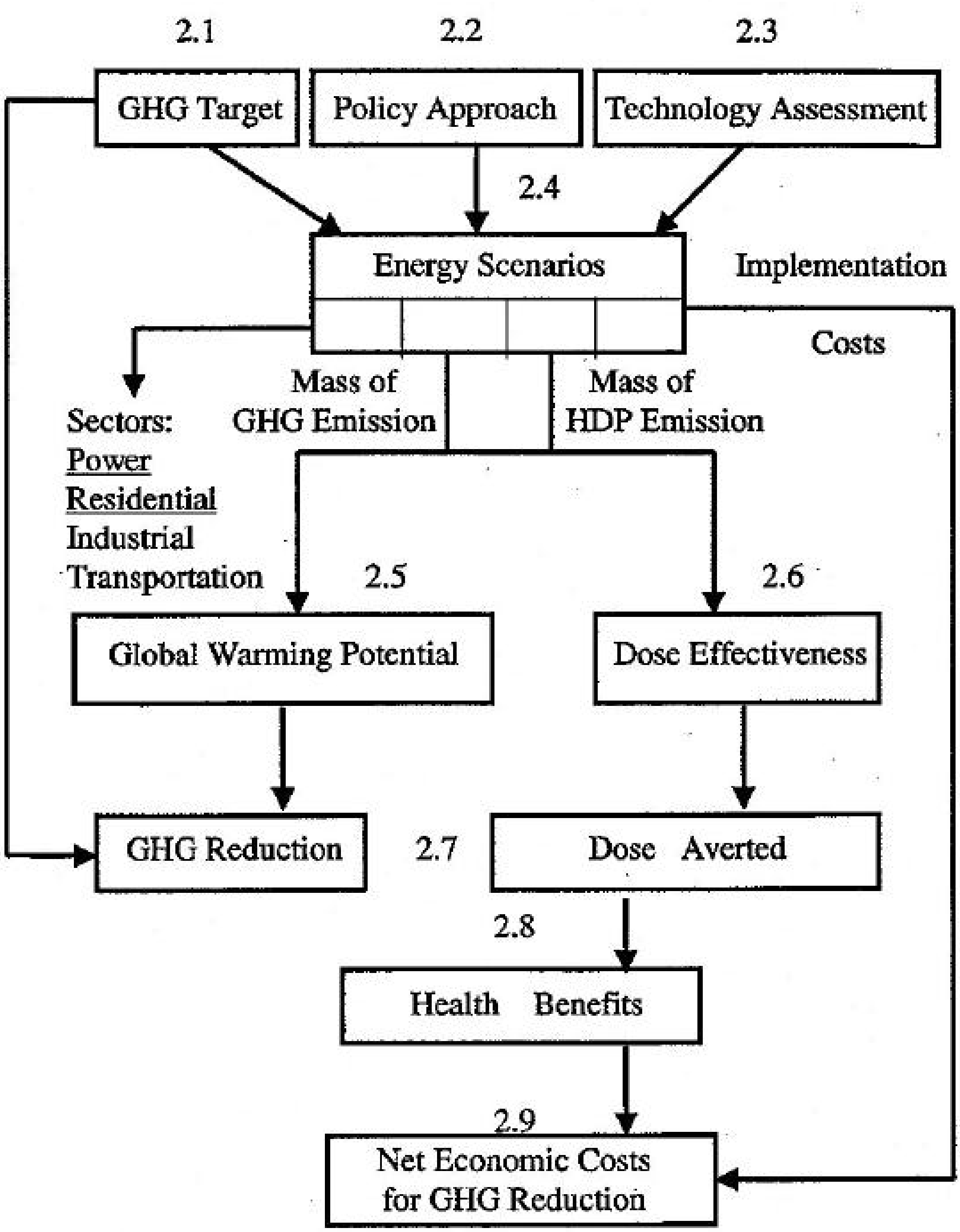
A rough indication of the potential scale of health benefits from GHG reduction can be had from estimates of the global burden of ill-health from air pollution today. Using airborne particulates as the indicator pollutant, WHO estimates that the global burden of premature mortality from air pollution is 2.7-3.0 million per year, or 5-6% of the global total (HESD, 1997).[1] Since most of this pollution comes from combustion of fossil and biomass fuels, which would be among the principal targets of any GHG control regime, the potential reduction in health-damaging emissions would seem to be at least as great as the target reduction in GHG emissions. Arguably it is even greater, however, since fuel switching from dirty, less efficient fuels such as coal to cleaner, more efficient fuels such as natural gas have even greater reductions of HDP emissions than they do of GHG emissions. With GHG reduction targets in the order of 10-20% below BAU, the scale of HDP emissions and associated reduction of ill-health could well be in the same range or somewhat higher, perhaps a 250,000 - 750,000 annual reduction in premature deaths worldwide.[2]

To more accurately estimate these near-term health benefits, it is necessary to link each specific technological option taken in a particular GHG reduction scenario with the accompanying HDP emissions reductions, as illustrated in Figure 2. The health impact of these emissions, however, is closely dependent on the sector of the economy in which they are taken. This is because the degree of human exposure created by a unit of HDP emissions depends on where they are released in relation to where people spend time, their <u>exposure or dose effectiveness</u> . Thus, a ton of emissions averted in the household sector close to where people live much of the time will generally cause a much greater reduction in human exposure than a ton averted in the industrial sector. As shown in the figure, once the change in exposures is determined, the resulting change in health risk can be calculated using exposure-response information from available, but still not definitive, epidemiological studies and other information.

[1] At 570 thousand, the preliminary estimate of the attributable burden of global deaths to air pollution in the Global Burden of Disease (GBD) database (Murray & Lopez, 1996) is substantially less, mainly because indoor air pollution was left out. The GBD is now being updated to remedy this omission.

[2] A preliminary estimate of a similar magnitude was recently published (Lancet, 1997). As it relied on models that essentially assumed that emission factors were equal in all regions and sectors, that emissions were spread equally across the entire land area of Earth, that biomass fuels did not contribute to emissions, and that indoor air pollution exposures were not counted, its result are difficult to interpret.

Figure 2. Analysis Framework



Numbers refer to sections in the text where each part of the analysis is discussed.

Here we apply this approach to the household and electric power sectors of China as a case study.[3] This serves not only to illustrate the methods, but because of the size of the Chinese population, the rate of its economic growth, and the extent of its dependence on dirty fuels, Chinese GHG emissions and air-pollution-related ill-health represent substantial fractions of the global total today and in the future. Even larger than US oil consumption, for example, Chinese coal represents the largest flow of fuel carbon in the world, in the sense of being under the influence of decisions by one government. The household and power sectors, in turn, represent around 60% of total energy demand[4] in China (Wang, 1997). Table 1 shows the energy use by energy sources in China's power and household sectors and total energy consumption in 1994.

Table 1 Energy Use by Energy Sources in China's Power and Household Sectors and Total Energy Consumption in 1994

Fuel	Power		Household[1]		Total	
	Mtce[2]	%	Mtce	%	Mtce	%
Coal	256	72.5%	163	39.5%	845	62.3%
Oil[3]	25	7.1%	1.6	0.4%	196	14%
Gas[4]	3.5	1%	16.7	4.1%	21	1.5%
Hydro	65	18%	0		65	4.8%
Nuclear	4.8	1.4%	0		4.8	0.4%
Biomass			230	56%	230	17%
Total	354[5]	100%	411	100%	1,357	100%

Sources: State Economic & Trade Commission (SETC), 1996; State Planning Commission (SPC), 1995; Wang & Fend, 1996; and Wang, 1997.

Notes:

1. Household energy use includes both urban and rural residential energy consumption.
2. Power generation (thermal, hydro, and nuclear power) in TWh is converted to energy consumption in Mtce on the basis of thermal power efficiency (380 gce/kWh, i.e. 32% efficiency) in China.
3. For the household sector, oil represents kerosene use.
4. For the household sector, gas includes natural gas, LPG, various forms of coal gas, and biogas. For the power sector and total energy use, gas indicates natural gas only, LPG is assigned to oil, coal gas to coal, and biogas to biomass.

[3] Most of the data on GHG and HDP emissions in China from technological options for energy efficiency and fuel switching used in this study derive from Wang, 1997.

[4] Total energy demand includes commercial fuels (coal, oil, gas) and power sources (hydro, nuclear, geothermal, wind) as well as the often non-commercial biomass fuels such as wood, crop residues, and dung, which are largely used in rural areas.

5. Among the total electricity generated, the household sector consumed 35 Mtce electricity.

China's annual economic growth averaged 11 percent from 1978 to 1993, and, as shown in Table 2, is expected to continue to 2020 at an annual average of 8 percent (World Bank, 1995). The future of the economy is promising, but in order to sustain the expected growth rates, China faces a number of challenges. One of the most important is reducing the adverse environmental impacts associated with rapid economic growth in a coal-based economy. Few countries in the world depend on coal as heavily as China does. Coal accounts for 90% of China's fossil fuel resources, and coal consumption reached 1.3 billion tons in 1995 (SETC, 1996), accounting for 80% of total primary commercial energy demand. China's CO_2 emissions reached 800 million tons of carbon in 1995, ranking second in the world, after the United States (Brown et al., 1997). China has 1.2 billion people, one-fifth of the world's population, who suffer as much as one-third of the global burden of air-pollution-associated disease (HESD, 1997; Florig, 1997). Table 3 shows that, under current trends, China will probably reach world average energy use and GHG emissions per capita by 2020. This means that China will be responsible for nearly a third of the global increase in GHG from 1990 to 2020.

Table 2 China's Estimated Population and GNP Growth

	GNP/capita	Growth rate	Population	Growth rate	Percentage of urban pop.
	US$/capita[1]	%/yr.	Billion	%/yr.	%
1990	340		1.14		26%
2000	800	9%	1.28	1.1%	31%
2010	1,725	8%	1.37	0.8%	37%
2020	3,000	6%	1.45	0.6%	42%

Source: World Bank, 1995.
Note :
1. RMB Yuan is converted to US dollars through the 1990 exchange rate: 4.78 Yuan/US$ (IMF, 1996).

Table 3 Estimated Increases in China's Commercial Energy Use and GHG Emissions Compared to Global Totals (without special commitment to GHG reduction Business as Usual Scenario)

Year	Total Energy Use Mtce		Per Capita Energy kgce		CO_2 Emissions GtC		Per Capita kgC	
	China	World	China	World	China	World	China	World
1990	950	11600	835	2190	0.6	6.0	523	1130
2020	3300	19700	2280	2520	2.0	10.9	1410	1400

Sources: World Bank, 1995; World Bank, 1994b; World Development Report, 1992; World Resources, 1996-97.

Thus, decisions about the course of China's GHG and HDP emissions reductions have important implications for a significant part of the world's population, as well as the world as a whole.

A full accounting of the near-term health benefits of GHG reductions would also need to include:

1. Changes in the minor GHG, in addition to the two major ones considered here;
2. Changes in the other important energy sectors: industry and transport;
3. Changes in non-energy sectors, for example, agricultural and forestry practices;
4. Changes in HDP other than the two major ones examined in this study;
5. Health impacts from changes in other factors besides HDP, for example, in occupational illnesses and accidents, waterborne diseases, and the risks of large accidents.

Although acknowledging that the analysis presented here falls short of such a complete evaluation, we believe that it does form a coherent and transparent framework into which these other facets can be added as desired. More practically, essentially all of the data needed to address issues 1-5, above, do not seem to be currently available in forms allowing them to be used directly for the purpose at hand. At the end of this report, therefore, we make recommendations for the type of additional research that would be needed to make future analyses more complete.

2. METHODOLOGY

The methodological framework followed in this analysis is shown in Figure 2. The first steps involve choosing realistic scenarios for GHG reduction. This involves:

Choice of a target level of GHG reduction, (see Section 2.1)and

Choice of the policy approach and energy sectors taken to achieve this reduction. Here we examine the two most likely approaches, energy efficiency (by better management and technology, accomplishing the same tasks with less energy) and fuel substitution (shifting from dirtier to cleaner fuels), in two sectors, household and power (2.2)

Combine with a set of technology assessments comparing the HDP emissions, GHG emissions, and economic costs of each major energy technology option on a per-unit-energy basis (2.3)

Determine reductions in the mass of HDP and GHG emissions (2.4)

Convert the GHG emissions reduction to overall Global Warming Potential (GWP) by appropriate weightings according to the action of each GHG in the atmosphere (2.5)

Convert reduction of HDP emissions to reduction of actual human inhaled doses based on inhaled dose effectiveness factors for each technology calculated from Chinese data (2.6)

Determine the total HDP-Dose reduction accomplished by meeting the GHG-GWP target (2.7)

The cumulative reduction in inhaled dose is used to estimate the cumulative reduction of human mortality and morbidity for each scenario using exposure-response relationships found in the current air pollution health literature (2.8)

Based on the health estimates,

Determine the marginal economic benefits of the reduced health effects associated with moving away from conventional coal use in the household and power sectors. From this, we calculate the marginal net economic costs of GHG reduction, which are the differences between the economic benefits from improved health and the incremental costs of the change in energy systems (2.9)

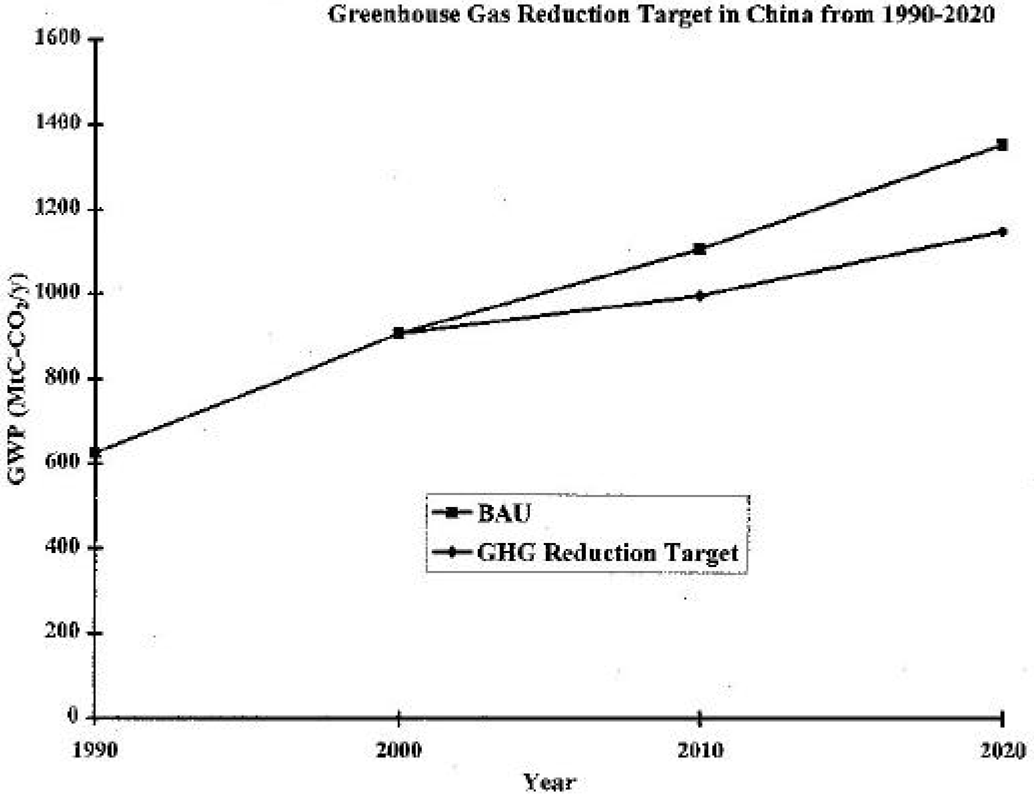
To provide a realistic baseline, all comparisons are made against a business-as-usual (BAU) scenario in which no special effort is made to control GHG emissions. The BAU does, however, account for reductions in HDP emissions that are expected to occur under current policies. Comparisons are made through the year 2020.

The final results are comparisons of the difference in health effects between BAU and two approaches to reach GHG reduction targets, Fuel Substitution and Energy Efficiency. The Energy Efficiency scenario maintains the same fuel mix as the BAU scenario, but accelerates the improvement in supply-side energy efficiency [1] to achieve the GHG reduction target. The Fuel Substitution approach is also examined under two least-cost scenarios, one in which the pathway follows the least-cost curve in attaining GWP reductions and the other in which the least-cost curve follows health-impact reductions. Finally, we present estimates of the marginal net economic costs of moving away from conventional coal use, considering the economic value of avoiding ill-health.

2.1 GHG Reduction Target

In keeping with typical targets used in international discussions of GHG control regimes (Lancet, 1997), we have chosen GHG reduction targets of 10% below BAU by 2010 and 15% below by 2020. Although a reduction, this represents about 60% above the 1990 level for China. It is what the US and Australia proposed for developing countries before the Kyoto meeting, but does not represent the official position of the Chinese government, which has not yet agreed to any reductions. Figure 3 shows the global warming potential from energy consumption for the BAU and our GHG reduction target in China from 1990-2020.

Figure 3. Global Warming Potential from Energy Consumption for the Business As Usual and Greenhouse Gas Reduction Target in China from 1990-2020



[1] Increases in supply-side energy efficiency can be achieved by adopting new efficient technologies, equipment, and/or improved management. Because the cost for energy efficiency improvement can be quite different from case to case, we did not estimate the cost of the Energy Efficiency scenario at this stage.

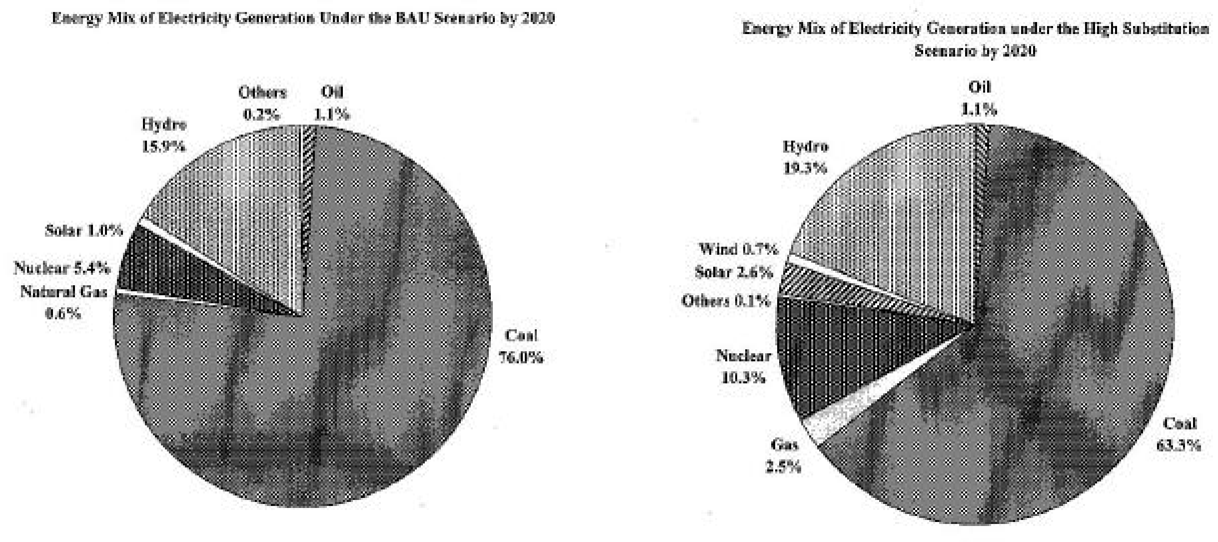
2.2 Policy Approach

Alternative future GHG control scenarios allow the policy makers to have better information on what could happen under different options. The emissions and energy projections in the scenarios are not forecasts, but, rather, estimates of what could occur under carefully and explicitly specified assumptions. Thus, the projections in this research are not intended to forecast the future, but to explore the human health benefits from different GHG reduction strategies.

We examined three energy scenarios: Business-As-Usual, Energy Efficiency, and Fuel Substitution Scenarios. Comparing alternative scenarios in the power and household sectors demonstrates that different pathways of technology and sector choices for GHG reduction could result in wholly different health benefits although achieving the same GHG reduction target. See Appendix A for details.

As much as possible, we adapted existing energy scenarios for China, including those of the World Bank, IPCC, US Environmental Protection Agency (EPA), and Lawrence Berkeley National Laboratory (LBNL). To estimate the human health implications, however, the energy scenarios need to be specific by sectors. We were surprised, however, to find out that most published scenarios did not provide sector details, and could not be directly used in our research (See Appendix B). It was necessary to make appropriate assumptions and conduct supplementary analysis to fill this gap. Figures 4 and 5 show the energy mix of the BAU and Substitution Scenarios[2] in the power and household sectors by 2020. Table 4 lists the average efficiency assumptions of coal-fired power plants under the BAU and Efficiency Scenarios [3]. The detailed assumptions are listed in Appendix A.

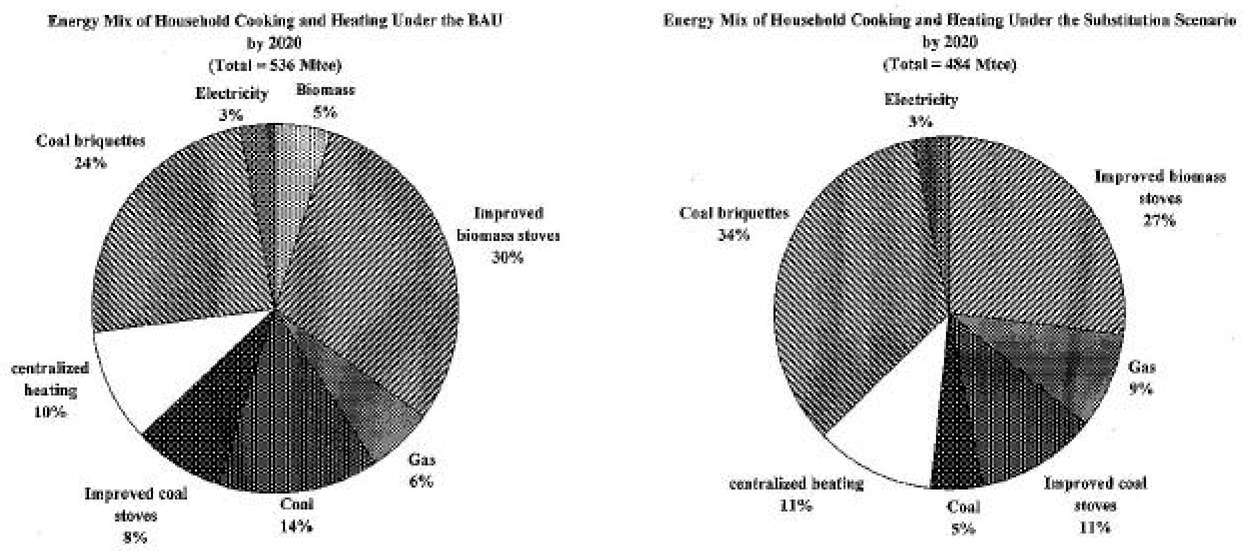
Figure 4. Energy Mix of Electricity Generation Under the BAU and Fuel Substitution Scenario by 2020



For Comparison, the total electricity generation in 1994 was 928 TWh

[2] The energy mix of the Fuel Substitution scenario is for the least-cost GWP reduction scenario.
3 The Efficiency Scenario in the household sector assumes that traditional coal stoves are replaced by improved coal stoves, coal briquettes, and centralized coal-fired heating facilities; and traditional biomass stoves are substituted by improved biomass stoves.

Figure 5. Energy Mix of Household Cooking and Heating Under the BAU and Substitution Scenarios by 2020



Total in 1990 = 429 Mtce, which was about 17% coal, 1% improved coal, 13% coal briquettes, 5% central heating, 1% gas, 19% biomass, 44% improved biomass

We chose the electricity and household sectors to conduct comparative assessment because these two sectors are not only important but also have quite different characteristics and thus well demonstrate the methods of comparative assessment. Power plants are generally centralized, while the residential fuel options are very decentralized. The emissions from the stacks of power plants are usually already controlled to some extent and although the electric power sector still emits a significant amount of pollutants, its contribution to ground-level pollutant concentrations is relatively small (Fang et al., 1995). On the other hand, the emissions from low-stack or no-stack household combustion devices are generally not subject to pollution control and are emitted indoors or outdoors at or near ground level in areas where people spend considerable time. The concepts, methods, and principles of the analysis can be illustrated fully in this fashion, although a full accounting of benefits should also include the transport and industry sectors.

Table 4 Efficiency[1] of Coal-fired Power Plants under the BAU and Efficiency Scenarios

		1990	2000	2010	2020
BAU	Kgce/kWh[2]	427	390	380	370
	%	28.1%	30.8%	31.6%	32.4%
Efficiency	Kgce/kWh	427	350	340	315
	%	28.1%	34.3%	35.3%	38.1%

Notes:
1. Efficiency is the average value for both old and new power plants.
2. kgce/kWh indicates coal consumption per kWh electricity generated. 1 kgce = 30 MJ

2.3 Technology Assessment

For our purposes, the comparative assessments of different energy technologies are best performed per unit of energy benefits (Holdren et al., 1980). This can be done by either comparing alternative ways to obtain the same energy form, e.g., liquid fuel, or alternative ways to supply the same final energy services, e.g., transport. In the power sector, comparative assessments are commonly based on a kWh of electricity delivered. Preferably, however, facilities providing the same kind of load, for example, peak or base, would be compared because there are of course differences between the value of electricity generated to meet base and peak loads. In the residential sector, where comparisons of energy alternatives involved different energy forms (solid, liquid, and gas fuels, electricity), the same final energy services are chosen as the denominators. Thus, the comparative assessment in the household sector is based on a delivered GJ heat supplied to the house in the heating system, and per delivered GJ heat into cookpot in the cooking system. Again, as there can sometimes be differences in heating rate, turn down ratio, and other factors not captured by simply delivered heat, it is best if efficiencies are measured for cooking the same mixture of foods.

2.3.1 Technology Choice

The health benefits associated with GHG reductions vary dramatically with the choice of technologies and sectors. Switching from conventional use of coal to various alternative energy choices, for example, can have different percentage of reductions in greenhouse gases and health-damaging pollutants, that is, different HDP/GHG ratios. Switching from coal-fired power technology to gas-fired power technology, for example, can substantially reduce health-related pollutants (e.g. particulates), but has limited GHG reductions compared to switching to hydropower, which would have significant benefits of both kinds.

In the power sector, coal, oil, natural gas, nuclear, hydro, solar, wind, and biogas power technologies are chosen for comparison. In China's residential sector, the existing energy options for urban and rural cooking include traditional biomass stoves, improved biomass stoves, traditional coal stoves, coal briquettes with improved coal stoves, gas stoves (coal gas, natural gas, LPG, and biogas), and electric stoves[4] (World Bank, 1994c; EWC et al., 1994; and Wang,

[4] Kerosene stoves are not commonly used in China and account for only about 0.5% of the total household energy use in China (Wang, 1997).

1997). The current energy options for urban and rural heating comprise traditional biomass stoves, improved biomass stoves, traditional coal stoves, coal briquettes with improved coal stoves, central heating, and district heating [5] (World Bank, 1994c; EWC et al., 1994; and Wang, 1997). Biomass fuels are examined on both a completely renewable basis and a completely non-renewable basis. [6] In both sectors, these sets of technologies represent realistic alternatives in the near future for China (Wang, 1997).[7]

2.3.2 GHG emissions

Although CO_2 is the principal and best-known greenhouse gas, it is by no means the only one. The same amount of carbon emitted in the form of methane, for example, creates many times the warming as it would as CO_2. Thus, devices with inefficient combustion, for example small stoves using solid fuels, in which a significant amount of fuel carbon is converted to products of incomplete combustion such as methane, can have quite different total global warming potentials if non-CO_2 gases are included (Smith 1994a). In addition, methane is released in other parts of fuel cycles, for example from coal mines or as leakage from gas pipelines. We thus account for emissions of these two most important GHGs: CO_2 and CH_4. Other greenhouse-related gases are also released from fuel combustion, including N_2O, CO, and non-methane hydrocarbons (NMHC). Since complete emissions data for N_2O were not available to us and the global warming potentials of CO and NMHC are site dependent, we did not include them at this stage. The other three major GHGs included in the Kyoto Protocol, HFCs, PFCs, and SF-6 are not generated from fuel combustion and are thus excluded from this research. In addition, primary and secondary particles also affect climate by changing albedo and influencing sunlight absorption in the atmosphere. We did not include the particle effects in this research.

2.3.3 HDP Emissions

We account only for particulates and sulfur oxide (indexed by SO_2) emissions from energy technologies, although recognizing that there are significant emissions of carbon monoxide, toxic organics, nitrogen oxides, and other HDP pollutants that could also be included in later versions of the study.

Many studies show that both short- and long-term changes in fine particulates are strongly associated with differences in human mortality and morbidity. Although there may be reasons to believe that part of this strong association may be due to the fact that particulates act as surrogates for total pollution or for some, as yet unknown, combination of pollutants, particulates seem to be the best single indicator of human health impacts from fuel combustion. We use them in this way here.

[5] These heating options are only permitted in the heating zone, which is primarily located in North, Northeast, and Northwest parts of China. The total population in the heating zone is 514 million people, or 45% of the nation' s total population (Wang, 1997). Heating is usually permitted between November 15 and March 15.

[6] With renewable harvesting, CO_2 is not counted as part of net GHG emissions, since it is recycled.

[7] Forthcoming databases based on more detailed GHG/HDP emissions from Chinese household stoves will soon enable refinement of these calculations Smith, et al. (1998).

Although sulfur oxides themselves have also been associated with ill-health, an effect separate from that of particulates has not been consistently demonstrated[8]. SO_2 emissions, however, do contribute directly to particulate levels by partly converting in the environment to solid sulfates and sulfites in the form of fine particulates that add to those released directly by the combustion device. Thus, not to ignore SO_2, we include an enhancement to particulate levels above that from particulate emissions alone in the form of a percentage of SO_2 emissions (see Appendix D). This we do only for outdoor emissions since conversion to sulfates does not have time to occur indoors.

2.3.4 Economic costs

In addition to the GHG and HDP emissions, the economic costs per unit energy output have been determined for each technology option in both sectors. Standard techniques using levelized costs and a 12% real discount rate were employed as described in Appendix C. Estimated market values of fuels were used instead of prices, many of which are subsidized in China. The result is provided in 1990 US dollars per kWh for the power sector and US dollars per GJ delivered cooking energy for the residential sector.

2.3.5 Boundary of Environmental Assessment

The environmental comparisons in this analysis address the entire fuel cycle or chain, as shown. Only the operational stage in each step is considered, however, and not the impacts from construction, dismantling, or long-term waste management.

Mining → Processing → Transporting → Conversion → T & D → End-use

Environmental comparisons in the electricity sector are conducted from fuel mining to transmission and distribution (T&D) along the fuel chain to calculate the damages per delivered kWh of electricity.

In the residential sector, environmental comparisons are accomplished from fuel mining to end-use along the fuel chain to estimate the damages per delivered GJ of useful heat. Hence, emissions from both the point of fuel production and fuel consumption are included in the household sector.

We examine <u>net</u> GHG emissions of the whole fuel chain rather than the <u>gross</u> releases. For example, if harvested renewably, biomass fuel does not generate net CO_2 emissions to the atmosphere, although if burned in inefficient devices it may produce net methane releases. In contrast, CO_2 emissions from net deforestation of a woodfuel source, which indicates that more trees were cut than grown, is counted. The net carbon emissions from biogas are the gross methane emissions from biogas leakage and combustion minus the avoided methane emissions from the decomposition of dung that would have occurred if it had not been used to make biogas, which assumes that the animals are fed on renewable fodder.

[8] Although studies in China sometimes show a significant association between SO_2 and daily mortality (Xu et al., 1994), most US studies suggested that associations between SO_2 and daily mortality are not statistically significant (Dockery et al., 1992; and Pope et al., 1992). It would be useful to examine this discrepancy in more detail in later studies.

2.4 Determine the Mass of GHG and HDP

The primary results of the technology assessments described above are summarized first in Figures 10 and 11. By sector, these place each technology according to the weight of GHG emissions and HDP emissions per unit energy benefit.

2.5 Conversion of GHG Emissions to Global Warming

The Global Warming Potential (GWP) is used in this study as the index to combine the impacts of different greenhouse gases. This is done by converting other greenhouse gases to CO_2 equivalent global warming effects per gram carbon. Used here is the 20-y GWP of methane, which is 25. This means that one gram of carbon as methane has the global warming effects of 25 grams of carbon as carbon dioxide over a 20-year period (IPCC, 1995).[9] Emissions of CH_4 in kilograms per kWh are converted to CO_2 equivalents by first converting to a carbon (molar) basis and then multiplying by the appropriate GWP for methane. Thus, the total GWP is the sum of the CO_2 emission factor and the CO_2 equivalent of CH_4.

2.6 Conversion of HDP Emissions to Human Doses

Direct comparison of HDP emissions of each technology can be misleading in demonstrating the human health impacts of different energy alternatives. Human health impacts are not directly related to emissions, but rather to the exposures or inhaled doses,[10] which indicate how many grams of pollutants actually reach the places where people are breathing over a certain period of time. For the same weight of emissions, the dose and exposure effectiveness (the fraction that actually reaches people' s breathing zones) can vary by many orders of magnitude (Smith, 1993). For example, in the USA, emissions indoors have a dose effectiveness some 1000 times that of typical power plants (Smith 1988). Thus, per unit of HDP emissions, the actual health impact will vary by a similar factor.

Therefore, when comparing technologies of different types, it is necessary to convert HDP emissions to exposures or inhaled doses to obtain a true idea of human health effects. Here we multiply particulate dose effectiveness (PDE), which is defined as grams of particulates inhaled by humans per ton emissions, by HDP emissions to obtain a dose measure that better reveals the potential health effects from fuel combustion. A different PDE is calculated for each major type of technology depending on its location relative to the surrounding population .[11]

For air pollution from coal-fired power plants, we used a Gaussian plume model with Chinese meteorological data to estimate the changes in particulate concentration resulting from marginal changes in emissions (similar to the approach in World Bank, 1994a). We developed an exposure-distance curve (See Figure 6), so that air pollution exposures out to 50 km boundary from each power plant are counted (See Appendix D). Figure 7 shows the calculation of indoor

[9] In the discussion section, the sensitivity of the results to changes in CH_4 GWP will be explored.
[10] Exposure = C x P x t; Dose = C x B x P x t; where C is exposure concentration; B is breathing rate; P is affected population; and t is the duration of exposure. See Appendix D.
[11] For calculation of health effects, however, we use the intermediate term, exposure concentration, which is what has been monitored in nearly all epidemiological studies. Since the exposure unit is so cumbersome, however, we use inhaled dose in the discussion and figures. They are simply linked together by the breathing rate. See Appendix D and Glossary.

and outdoor exposures from power plants emissions. Appendix D explains the detailed method to calculate exposed population.

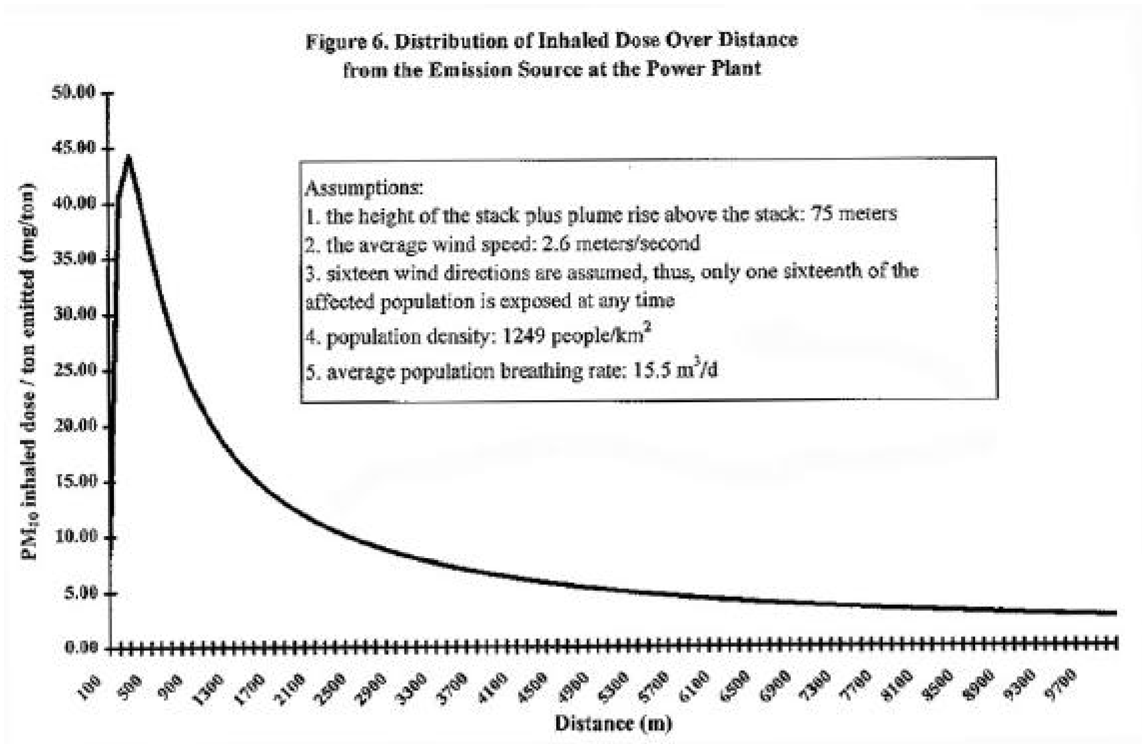


Figure 6. Distribution of Inhaled Dose Over Distance from the Emission Source at the Power Plant

Figure 7. Exposure from power plants emissions

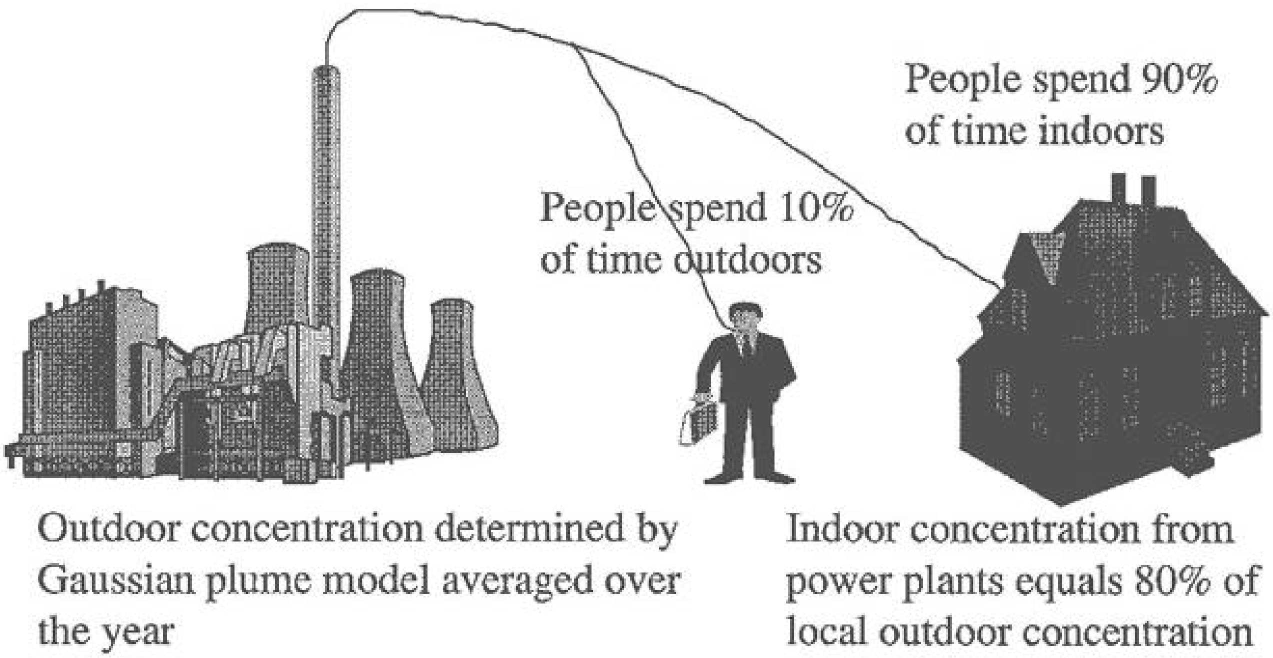


Figure 8 Exposure from household stoves emissions

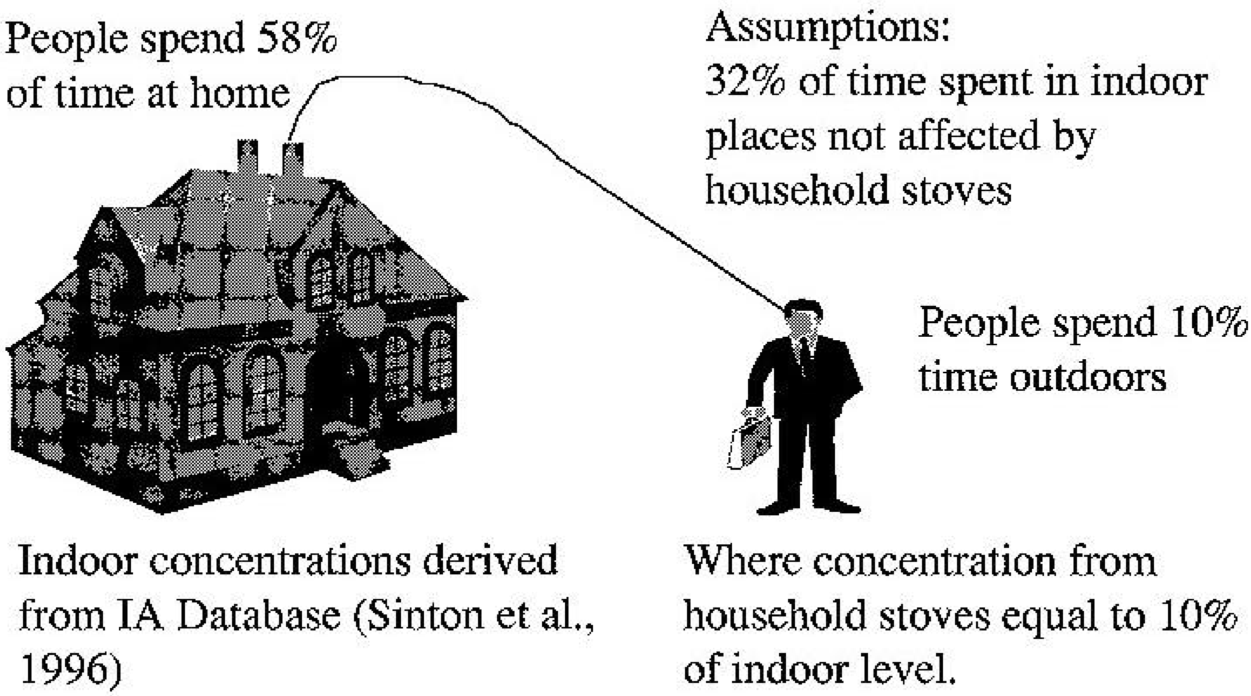


Figure 9. Comparison of Health Effects of One Ton Annual Particulate Emissions from Power Plants Vs. Household Stoves

Power Plants	**1 ton PM10**	**peak-1 µg/m^3 mean-0.037ng/m^3**	**265 (µg/m^3)-person-y**	**1.5 g**	**0.002 death**
Household Stoves	**1 ton PM10**	**peak-5000 µg/m^3 mean-150ng/m^3**	**12,000 (µg/m^3)-person-y**	**68 g**	**0.08 death**
Quantity and quality of fuel gives some idea of potential harm.	Emissions of air pollutants depend on how much of which type is burned in what way.	The concentration of air pollutants in the air depends not only on the emissions but also on the atmospheric conditions (or ventilation conditions inside a building if the concern is indoor pollution).	Exposure depends on how many people breathe the mean concentration and for how long.	Dose measures how much pollutant is actually deposited in the body and depends not only on exposure but also factors such as the rate of breathing and the size of the particles.	Health effects depend not only on dose but also on factors such as age, sex, whether the person smokes, and the existence of other diseases.

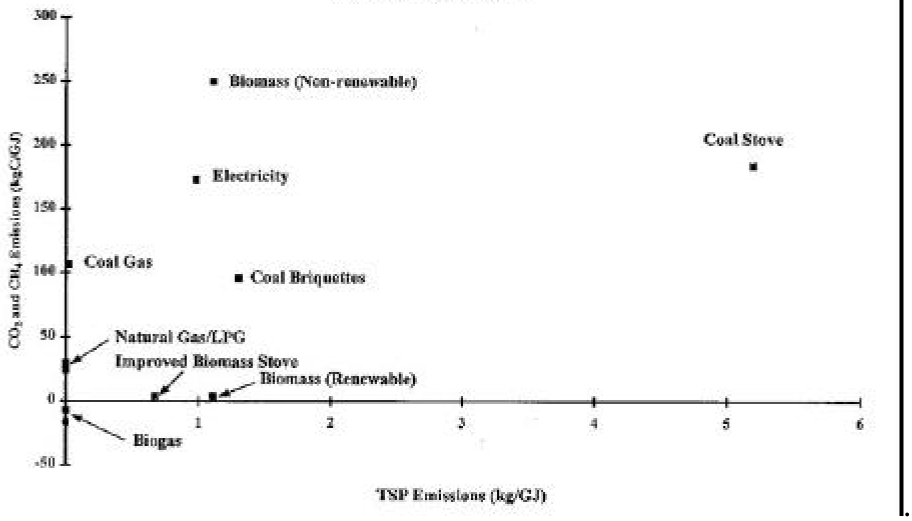
Measurement and control can be initiated at any stage

Indoor and outdoor exposures from household stoves are also determined, as shown in Figure 8. Outdoor concentration is assumed to be 10% of indoor level. This could be improved if better information becomes available [12] about ground-level "neighborhood" pollution from residential/commercial sources. The method is detailed in Appendix D. Results are summarized in Figure 9.

2.7 GWP/HDP Relationships for Major Technology Options

Figures 12 and 13 present the same relationships as Figures 10 and 11, but with GHG and HDP emissions by mass translated to actual warming capability (by multiplying GHG by GWP) and actual health damage potential, i.e. inhaled dose, (by multiplying HDP by PDE). These are the central results used throughout the rest of the study. Note, that there are substantial differences in the relative and absolute sizes of the global warming and health implications of technologies when they are weighted by GWP and PDE as is seen by comparing the relative positions of different technologies in Figures 10 and 11 with those in 12 and 13

Figure 10. Particulate Emissions vs. Greenhouse Gas Emissions of Household Cooking Options



[12] Several studies in China showed that indoor residential coal use contributes to about 60-80% of outdoor ground-level particulates and SO_2 concentrations (Tsinghua University, 1991; and Wang, 1993).

Figure 11. Particulate Emissions vs. Greenhouse Gas Emissions of Electricity Generation Technologies

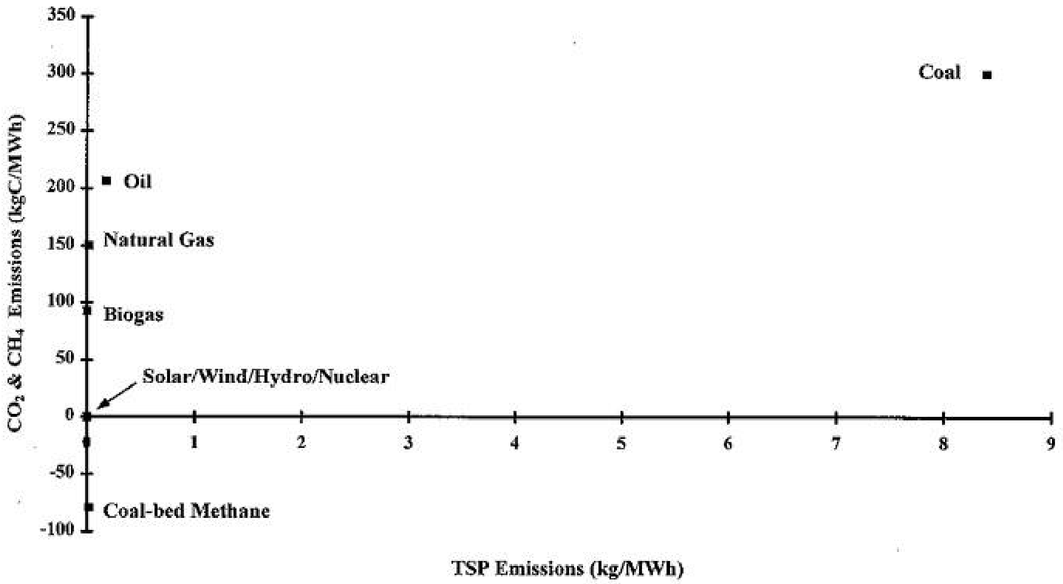


Figure 12. Human Health Impacts vs. Global Warming Effects of Household Cooking Options

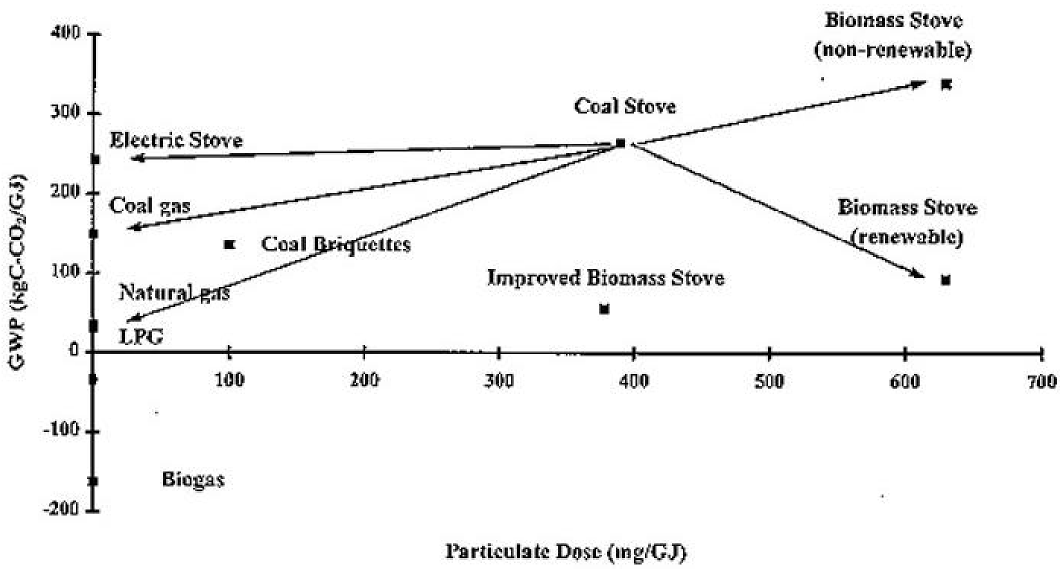
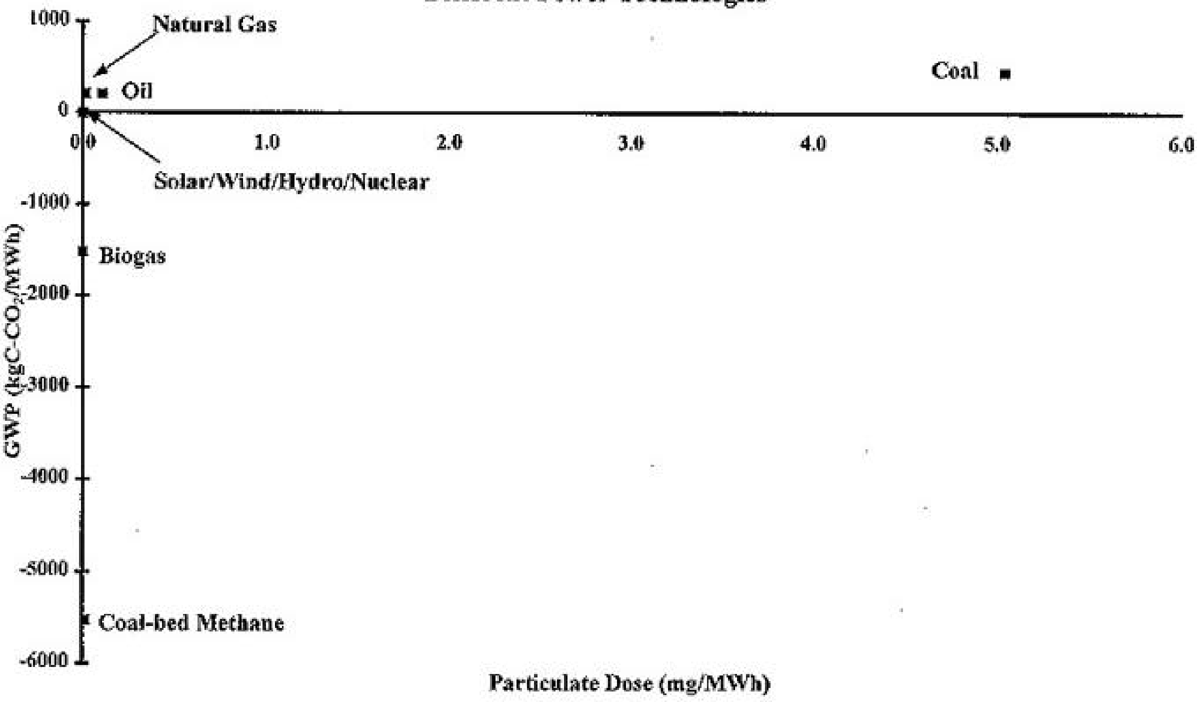


Figure 13. Human Health Impacts vs. Global Warming Effects of Different Power Technologies



The GWP and inhaled dose created by any particular mixture of these technologies can be determined from these figures by simply multiplying by the total energy production of each technology[13]. Note that there is a substantial difference in the location of technologies in the figures indicating that there is considerable difference in the HDP/GHG and dose/GWP ratios of different mixes of technologies. Furthermore, the dose and GWP implications of technological shifts can easily be derived from these figures. Symbolically such a shift is an arrow (vector) from one technology to the other. As shown in Figure 12, for example, a shift from traditional coal stoves to electric stoves results in a substantial decrease in particulate dose, but limited reduction in GWP.

Since the physical limits (only so much land is available for biomass production, for example) and economic costs have also been determined for each of these technologies, it is possible to derive least-cost development pathways in each sector. A least-cost supply curve looks like a series of rising steps. Each step on the curve represents one technology. Its width indicates the potential contribution of that technology to the supply mix in a given time frame, and its height indicates the costs per unit of energy provided by the technology. Thus, a mix of technology to meet energy requirements can be identified by "climbing the cost-supply staircase for energy" until the demand is reached (Wang, 1997). It is traditionally employed to compare the costs of different technologies and their potentials for meeting the demand for a particular energy service.

Three types of least-cost supply curves are most clearly of interest:

A. least-cost by economic cost per unit energy only
B. least-cost per unit GWP reduction scenario
C. least-cost per unit dose reduction scenario

Pathway A is essentially Business as Usual (BAU), although current energy trends are not entirely determined by cost considerations alone. Pathway B would be the optimum GHG reduction approach to reach the GWP reduction target at lowest economic cost. Pathway C would achieve health benefits at the lowest cost while achieving GHG reductions target as well. The net differences in GWP emissions and dose between Pathways A & B and A & C, in which B & C each are followed up to the GHG reduction targets, are evaluated in this report for the Fuel Substitution approach, making a total of three alternative scenarios in all. Figures 14 - 16 show the least-cost supply curves for pathway A - C respectively for the household cooking options by 2020.

[13] For example, the GWP of each household cooking option in Figure 12 is expressed in terms of kg carbon as CO_2 per GJ energy (kgC-CO_2/GJ). Thus, the total GWP of one particular energy option can be determined by multiplying the total energy production (GJ) by the GWP per unit energy (kgC-CO_2/GJ).

Figure 14. Least-Cost Supply Curve for Household Cooking Options Under the BAU Scenario by 2020

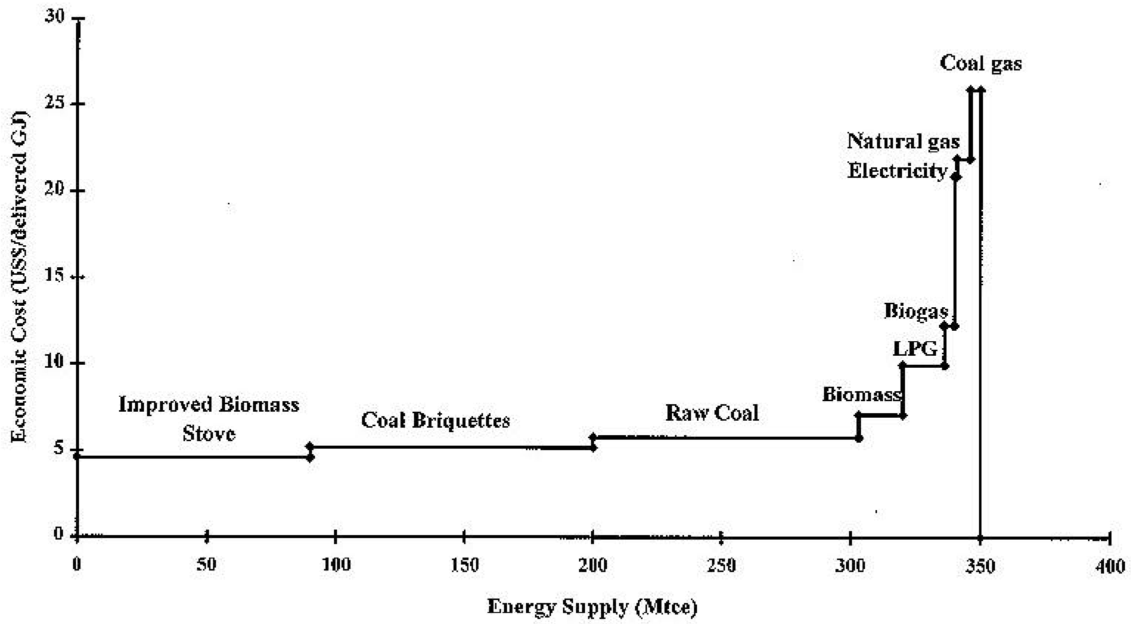


Figure 15. Least-Cost GWP Reduction Curve for Household Cooking Options by 2020

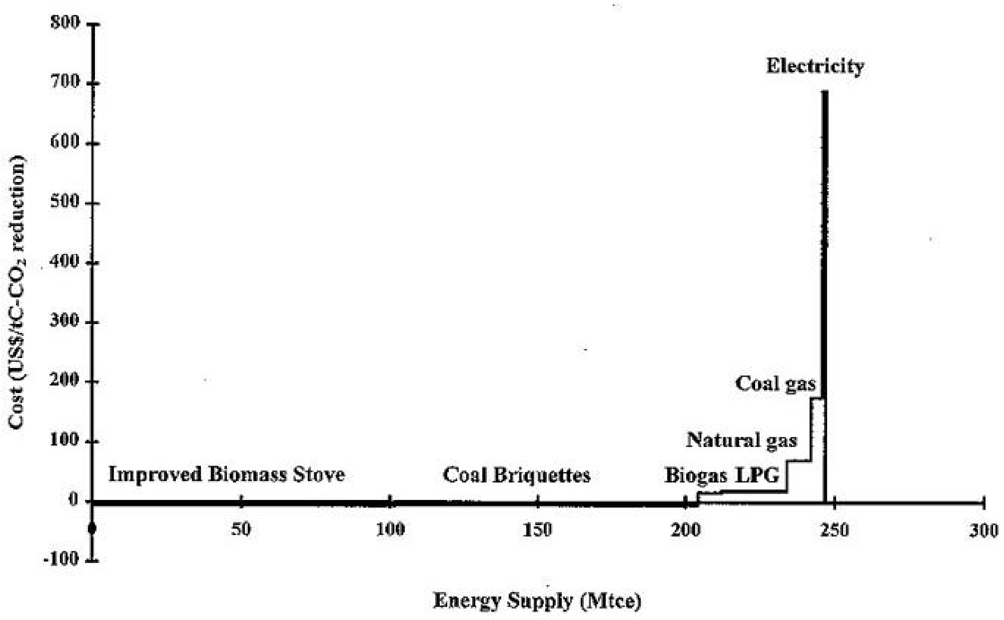
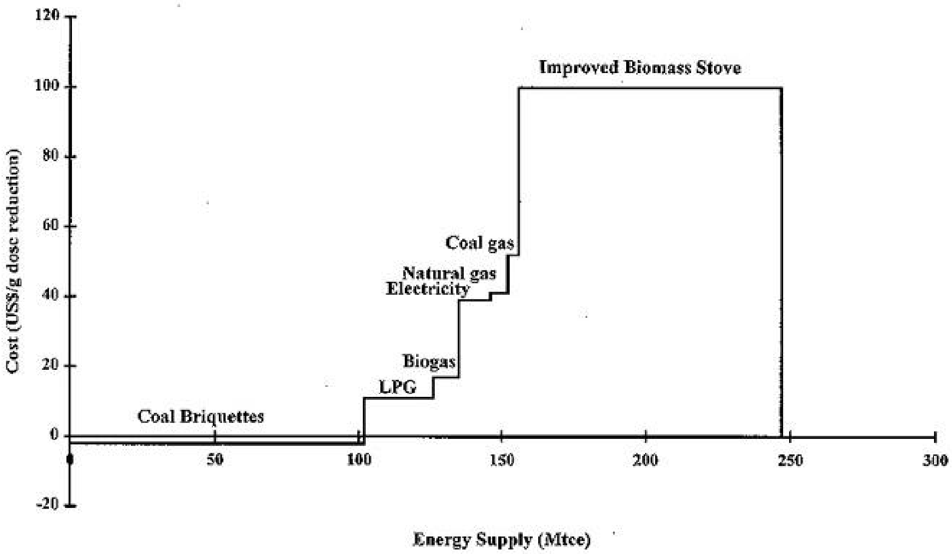


Figure 16. Least-Cost Dose Reduction Curve for Household Cooking Options by 2020



2.8 Conversion of cumulative inhaled dose reductions to health benefits

The next step is to convert the calculated levels of averted dose into equivalent averted health effects. In Appendix E, there is a brief review of some of the recent major epidemiological studies in Europe, North America, and China that have derived exposure-response relationships for short or long-term exposures to particulates.

For mortality, we use the Chinese estimates in Beijing and Shenyang[14] (World Bank, 1994a) as the lower end of the exposure-response relationship range, and the chronic exposure-response relationship in the cross-sectional studies in the US (Pope et al., 1996) as the higher end of the range[15]. A number of daily time-series studies in the US have remarkably consistent results, which are intermediate (Dockery et al., 1996). Thus, we use them to define the best estimate value.

Epidemiologic studies in developed countries also report reasonably consistent associations between PM_{10} and several morbidity outcomes. Ostro (1996) reviewed and summarized the morbidity exposure-response coefficient from existing studies, and gave a low, central, and high estimate for each morbidity endpoint, as listed in Table 5. The morbidity health endpoints include respiratory hospital admissions, emergency room visits, restricted activity days [16], lower respiratory illness in children, asthma exacerbation, respiratory symptoms, and chronic bronchitis.

[14] The exposure levels to indoor air pollution resulting from stove combustion in rural areas are much higher than those to the ambient air pollution in urban areas, thus, using exposure-response coefficient derived from the two cities may not be as representative of the health consequences of rural indoor air pollution .

[15] Particulate exposure in China is several times higher than the US. When we extrapolate the US exposure-response relationship to China, we assume a linear relationship.

[16] Restricted activity days mean days spent in bed, days missed from work, and other days when activities are significantly restricted due to illness. This endpoint is applied to the population above age 16.

Table 5 Exposure-Response Coefficients

Health Endpoints	Reference	Coefficient Estimate[1] Low	Central[2]	High
Mortality	Xu et al. (1994); Pope et al., (1996); Dockery et al. (1996)	0.04	0.1	0.3
Respiratory Hospital admission	Pope (1991)	0.7×10^{-5}	1.2×10^{-5}	1.6×10^{-5}
Emergency room visits	Samet et al. (1981)	13×10^{-4}	24×10^{-4}	34×10^{-4}
Restricted activity days for adults above age 16	Ostro (1990)	0.04	0.06	0.09
Acute Bronchitis for Children below age 16	Dockery et al. (1989)	0.8×10^{-3}	1.6×10^{-3}	2.4×10^{-3}
Asthma attacks per asthmatic	Ostro et al. (1991) Whittemore & Korn (1980)	0.03	0.06	0.2
Respiratory symptoms	Krupnick et al. (1990)	0.09	0.18	0.27
Chronic Bronchitis for adult above age 16	Abbey et al. (1993)	3×10^{-5}	6×10^{-5}	9×10^{-5}

Sources: Ostro, 1996; Pope et al., 1996; Dockery et al., 1996; and Xu et al., 1994.
Notes:

1. All functions are for the entire population except as noted. The coefficient for mortality is expressed in mortality percent increase per ug/m^3 change in PM_{10}, while the coefficient for morbidity is in terms of annual morbidity cases per ug/m^3 annual average change in PM_{10}.
2. The central value for mortality is the best estimate value, as explained in the text. The central value for morbidity is the arithmetic average of the low and high values.

Then, the change in mortality attributable to the reduced particulate emissions is a function of the exposure-response coefficient (in terms of percent effect per ug/m^3), the change in particulate concentrations, crude death rate, and the size of the affected population[17], as shown in the following formula (Ostro, 1996).

$$\begin{aligned}\text{Avoided mortality} = {} & \text{percent effect per } ug/m^3 \\ & \times \text{ baseline mortality rate}^{18} \\ & \times (1/100) \\ & \times \text{ change in } PM_{10} \\ & \times \text{ exposed population}\end{aligned}$$

[17] Changes in exposures = changes in PM_{10} concentration / ton particulate emission x exposed population (See Appendix D) x total energy production (GJ) x particulate emission factor (ton/GJ).

[18] The annual crude death rate for China in the mid-1990s is about 0.007 (WHO, World Health Report, 1997).

The avoided morbidity is calculated by the exposure-response coefficient (in terms of morbidity cases per ug/m^3) multiplied by the change in PM_{10} and exposed population (Ostro, 1996).

$$\text{Avoided morbidity} = \text{morbidity cases per } ug/m^3 \times \text{change in } PM_{10} \times \text{exposed population}$$

In this way, we obtain the total avoided mortality and morbidity from the alternative scenarios over the BAU scenario in the power and residential sectors.

2.9 Net Economic Costs for GHG Reduction

Shifting from high-carbon fuel/technology to low-carbon fuel/technology to reduce GHG emissions is usually accompanied by increasing economic costs. As mentioned earlier, however, such a fuel/technology shift will not only reduce GHG emissions, but also abate the emissions of health-related pollutants such as particulates. Thus, the economic benefits of improved human health impacts generated from the fuel/technology switch should also be counted. The net economic cost, which is calculated by deducting the economic benefit of improved health impacts from the incremental economic cost associated with the fuel/technology switching, can be used to measure the real economic cost (or benefit) of GHG reduction.

There is no universal or universally accepted way to convert improved health impacts into economic benefits, but rather several possible approaches. This study adopted the World Bank estimate (1994a) of China' s willingness to pay (WTP) for a reduction in risk of premature death for the economic benefits of avoided mortality, and the wages and the costs of medical treatment for the economic benefits of avoided morbidity. We only compared the net economic costs vs. the economic benefits of GHG reduction per unit of energy when changing away from conventional coal use to other fuels and/or technologies. Since the net economic costs for technology changes are those under current technology development, the economic benefits are based on current WTP[19]. See Appendix F for details. Figures 17 and 18 show the results in the household sector and power sector respectively. A more detailed examination of such costs in Chinese conditions would be a valuable supplement to this work.

[19] If the future investment costs are involved, then the economic benefits should be based on future WTP, which will be much higher than today' s WTP given the high income growth rate in China.

Figure 17. Net Economic Costs vs. GWP Reduction of Changing from Coal Stove to Other Household Cooking Options

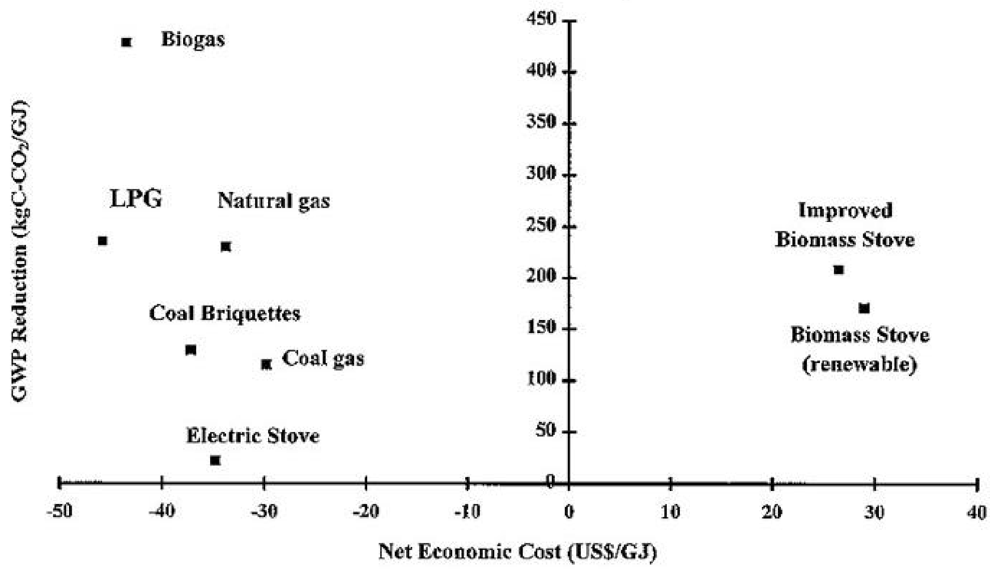
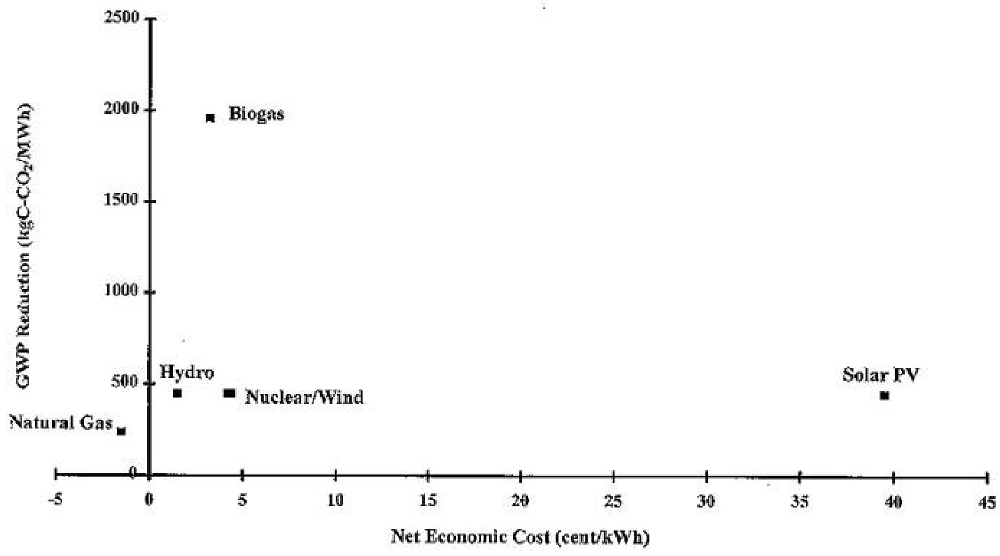


Figure 18. Net Economic Costs vs. GWP Reduction of Shifting from Coal Power Plants to Other Power Generation Technologies



3. RESULTS

Following the methodology described above, we first analyzed the human health benefits vs. GHG reduction of two types of technological change within each of the two sectors. Then, the health benefits associated with change in energy use in the power sector are compared with those in the household sector. From these comparisons, we present the results of the avoided mortality and morbidity of the three alternative scenarios relative to the BAU case (two different least-cost Fuel Substitution scenarios and one Energy Efficiency scenario). Based on the health estimates, the marginal net economic costs (or benefits) of reducing GHG emissions by moving away from conventional coal use are determined.

3.1 Human Health Benefits vs. GHG Reduction

As shown in Figure 12, if traditional coal stoves shift to coal gas, the human health benefits are much larger than the GWP reduction. If traditional coal stoves switch to natural gas/LPG, there would be almost proportional benefits of human health and GWP reduction. On the other hand, if traditional coal stoves change to non-renewable traditional biomass stoves [1], both the human health impacts and global warming effects would become worse. If traditional coal stoves shift to renewable biomass stoves, the GWP reduction is much larger than the human health benefits. This difference is because biomass fuels do not generate net CO_2 buildup in the atmosphere when operated on a completely renewable basis. Biomass stove combustion, however, may make a net contribution of other GHGs such as CH_4 (N_2O, NMHC, and CO) due to incomplete combustion (Smith, 1994a).

As shown in Figure 13, if conventional coal-fired power plants shift to natural gas-fired power plants, the human health benefit, measured by particulate dose, is larger than the GWP reduction. On the other hand, if conventional coal-fired power plants shift to hydro power or other renewable power plants, the human health benefits are in direct proportion to GWP reduction. Therefore, different technology choices achieve different degrees of health benefit in the course of attaining the same GWP reduction.

3.2 Sectoral Assessment

As just shown, the health benefits associated with changes in energy use vary greatly with technology choices in the same sector for the same GHG reduction. Between sectors, however, the variation is even larger.

As shown in Figure 9, one ton of particulate emission from power plants can result in 1.5 grams of PM_{10} being breathed by humans and 0.002 deaths. On the other hand, one ton of particulate emission from household coal stoves can result in 68 grams of PM_{10} being inhaled by people and 0.08 deaths. Therefore, under the conservative assumptions made here about indoor exposures, the human health benefits of one ton reduction in particulate emissions from household coal stoves are at least 40 times larger than those from coal-fired power plants[2].

[1] Non-renewable biomass fuel means that the new growth ~~can~~does not replace the harvested materials.
[2] In some extreme cases, where there is no chimney in the kitchen, daily indoor particulate concentration can be as high as 1000-2000 ug/m^3 (Sinton et al., 1996). Under such a case, the health benefits of marginal reduction in

Therefore, GHG reductions resulting from changes in energy use are generally accompanied by improved human health benefits, however, the choices of energy technologies and sectors determine the health benefits for the same GHG reductions.

3.3 Health Benefits of the Alternative Scenarios

Tables 6 and 7 list the health benefits of the three alternative GHG energy scenarios in the power and household sector. Although the three alternative scenarios can achieve the same GHG reduction target, the health benefits are quite different. To some readers, the numbers of averted premature deaths may appear large (75-123 thousand per year for 2010 by best estimate). They represent, however, only about 10% of the total premature deaths from air pollution estimated for the early 1990s in China (Florig, 1997). They indicate a potential reduction of 1-4% in total mortality by 2020.

Table 6 Annual Avoided Death for Different Scenarios in the Power and Residential Sectors

Year	Scenario	Sector	Low	High	Best
2010	Efficiency	Power	700	6,000	2,000
		Household	43,000	320,000	110,000
	Substitution				
	least-cost GWP	Power	850	7,200	2,400
		Household	29,000	220,000	73,000
	least-cost dose	Power	900	7,700	2,600
		Household	48,000	360,000	120,000
2020	Efficiency	Power	1,500	13,000	4,400
		Household	62,000	460,000	150,000
	Substitution				
	least-cost GWP	Power	1,700	15,000	5,000
		Household	47,000	360,000	120,000
	least-cost dose	Power	1,800	16,000	5,200
		Household	70,000	530,000	180,000

Note: The Global Burden of Disease database predicted that the total mortality in China will reach 12 million people by 2010 and 14 million people by 2020, and the total population in China is projected to be 1.38 billion by 2010 and 1.47 billion by 2020 (Murray & Lopez, 1996).

In the power sector, the Substitution Scenario has higher health benefits than the Efficiency Scenario. Increasing the efficiency of coal-fired power plants results in the same percentage reduction in both GWP and particulate doses. On the other hand, shifting from coal to natural gas power generation technologies, for example, has higher reduction in particulate doses than GWP (See Figure 13). Thus, for the same GHG reduction, fuel substitution strategy can result in more avoided mortality and morbidity than increase in efficiency in the power sector.

In the household sector, however, the Efficiency Scenario has higher health benefits than the Substitution Scenario (least-cost per unit of GWP reduction scenario).

particulate emissions from household stoves could be 500-1000 times larger than those from power plants (Smith, 1993).

Table 7 Annual Avoided Morbidity for Different Scenarios in the Power and Residential Sectors by 2020

Scenario	Sector	Morbidity	Low	Central	High
Efficiency	Power	Respiratory hospital admissions	3,800	7,000	9,000
		Emergency room visits	750,000	1,400,000	2,000,000
		Restricted activity days	16,000,000	23,000,000	37,000,000
	Household	Respiratory hospital admissions	140,000	260,000	340,000
		Emergency room visits	28,000,000	52,000,000	75,000,000
		Restricted activity days	620,000,000	880,000,000	1,400,000,000
Substitution	Power	Respiratory hospital admissions	4,300	7,800	10,000
Least-cost GWP		Emergency room visits	830,000	1,500,000	2,200,000
		Restricted activity days	18,000,000	26,000,000	41,000,000
	Household	Respiratory hospital admissions	110,000	200,000	260,000
		Emergency room visits	22,000,000	40,000,000	58,000,000
		Restricted activity days	470,000,000	680,000,000	1,000,000,000
Substitution	Power	Respiratory hospital admissions	4,500	8,000	11,000
Least-cost dose		Emergency room visits	890,000	1,600,000	2,400,000
		Restricted activity days	19,000,000	28,000,000	43,000,000
	Household	Respiratory hospital admissions	130,000	240,000	320,000
		Emergency room visits	26,000,000	48,000,000	69,000,000
		Restricted activity days	700,000,000	1,000,000,000	1,600,000,000

Increasing efficiency in the household sector involves shifting from traditional coal use to improved coal stoves and coal briquettes. As shown in Figure 12, this switching process results in more reduction in particulate doses than GWP. Under the Substitution scenario, however, traditional coal use changes to natural gas and LPG. Compared to the energy efficiency strategy, such a shifting has similar reduction in both particulate doses and GWP (See Figure 12). Hence, increase in energy efficiency in the household sector can result in more avoided mortality and morbidity than fuel substitution strategy, to achieve the same GHG reduction target.

In both the power and household sectors, the least-cost per unit dose reduction scenario has higher health benefits than the least-cost per unit GWP reduction. When shifting away from conventional use of coal, there are different combinations of fuel choices to achieve the same GHG reduction target. When shifting from traditional coal stoves to electric stoves, for example, reduction in particulate doses is much more than the GWP reduction (See Figure 12). As shown in Figure 15, an electric stove is the most expensive option to reduce GWP, while its unit cost for dose reduction is not high (See Figure 16). Thus, the percentage of electric stoves in the least-cost per unit dose reduction scenario is much higher than that in the least-cost per unit GWP scenario. Hence, different choices of alternative fuels have different health benefits, although the same GHG reduction target is achieved.

In addition, the human health benefits resulting from changes in energy use in the household sector are at least 40 times larger than those in the electric power sector, although the same GHG reduction goal is met in both sectors. This illustrates that different energy technology and sector choices can have different health benefits with the same GHG reduction. Therefore, choice of energy technologies and sectors is more important than the simple target of GHG reduction for improved public health benefits.

The Substitution Scenario has more potential to reduce GHG emissions in the long term than the energy efficiency strategy. The maximum level of GWP reduction by the fuel substitution strategy can reach around 30% below BAU by 2020 in both the power and residential sectors, while the energy efficiency strategy can reach only 15% reduction in GWP below BAU by 2020. Combining the Efficiency and Substitution scenarios together, however, will result in less than the simple total of 40% reduction in GWP below BAU by 2020, because accomplishment of one will reduce the total available reduction for the other.

3.4 Net Economic Costs vs. GHG Reduction

The cost of GHG reduction alone, without considering any offset from health benefits, averages roughly about $40 per ton carbon (tC) (as CO_2) in the power sector and $5/tC in the household sectors. These compare favorably with costs of projects being considered in other parts of the world. The Global Environment Facility, for example, uses $10 tC as a guideline for many projects (GEF, 1996). With consideration of the health benefits, however, the costs are even more attractive.

As shown in Figure 17, when traditional coal use shifts to other household energy options for GHG reduction, the incremental costs are less than the economic benefits of improved health impacts in most cases, therefore, the net economic costs are negative (i.e., there is a net benefit). This demonstrates that when economic benefits of improved health impacts are counted, GHG reduction in the household sector can be achieved with a net economic gain.

Compared to Figure 17, Figure 18 demonstrates a quite different picture. When coal power plants switch to other alternative power technologies, the incremental economic costs are more than the economic benefits of improved health impacts in most cases, hence, the net economic costs are positive. This illustrates that the economic benefits of improved health impacts in the electric power sector is not large enough to offset the incremental economic costs associated with GHG reductions.[3]

[3] Elkins (1996) showed that, under Western European conditions, the economic value of health and other secondary benefits of CO_2 emission reductions could also offset much if not all abatement costs.

4. Recommendations for Further Study

This study was hampered by lack of data and information of several kinds. Future studies might well consider adding in some or all of the following components:

A. A full accounting of the near-term health benefits of GHG reductions might also consider including:

1. Changes in the minor GHG, such as nitrous oxide, in addition to the two major ones considered here, although the improvement in accuracy would likely be marginal;
2. Changes in the other important energy sectors: industry and transport, which would also need to consider the large range of exposure effectiveness of different technologies in these sectors;
3. Changes in non-energy sectors, for example, agricultural and forestry practices;
4. Changes in HDP other than the two major ones examined in this study, for example, carbon monoxide, nitrogen oxides, and ozone;
5. Health endpoints from air pollution exposures not considered here, in particular lung cancer, acute carbon monoxide poisoning, and the range of effects found from arsenic and fluorine exposures due to indoor coal combustion in China (Smith & Liu, 1994);
6. Health impacts from changes in other factors besides HDP, for example, in occupational illnesses and accidents, waterborne diseases, and the risks of large accidents. For example, as shown in Figure 4, the fuel substitution scenario in the power sector increases the fraction of electricity generated by nuclear and hydro, both of which are relatively clean from an air pollution standpoint, but have risks of large accidents and other potential problems.
7. More sophisticated health information might be included, for example lost-life years and disability- or quality-adjusted life-years lost.

B. More sophisticated energy/economic analyses might be performed that included changes in WTP over time, GWP and HDP income elasticity, and so on.

C. More detailed sector-level analyses are needed for all energy and GHG scenarios.

D. There is a clear need to improve emissions databases of both GHG and HDP for a range of current and future technologies and use patterns.

E. There is need to conduct more systematic monitoring of indoor air quality throughout China, particularly in rural households using coal and biomass fuels. The indoor concentrations used here, although at the lower end of the range reported in the Indoor Air Pollution Database for China (Sinton et al., 1996), may have been lowered even further in recent years by the large rural improved stove program and other factors (Smith, et al. 1993). If this were found to be the case, the total scale of the health benefit from increased efficiency and fuel switching in the household sector would be lowered, although not the conclusion that the marginal benefit of control is much higher than in the power sector.

F. There is also a need to evaluate more thoroughly the population dose effectiveness (PDE) from representative power plants and other classes of facilities in China to pinpoint the relationship between emissions and ill-health more accurately than we have done here.

G. Other non-health near-term benefits from GHG control might be included, such as ecosystem protection, energy security, etc.

H. Because China is so large and diverse in population, economy, and geography, it will be necessary to disaggregate this kind of analysis at least to the regional level before concrete policies can be formulated.

I. There is an urgent need to increase the quantity and quality of information linking emissions, ambient concentrations, and true human exposures related to both indoor and outdoor pollution sources in China.

J. Similarly, the current level of information on exposure-response relationships is not commensurate with the apparently immense scale of human health impact involved. The wide ranges in our reported results largely reflect the uncertainties in I and J.

5. Discussion and Conclusions

Recognizing the uncertainties in the data and methods outlined just above, several tentative conclusions can be drawn:

- Many GHG reduction strategies will have human health benefits from HDP reduction, but the size of the health benefits vary greatly with energy technologies and sectors.
- Choices of energy technologies and sectors are more important than choice of the exact target of GHG reduction in determining improved public health.
- When all of the three alternative scenarios (energy efficiency, least-cost GWP reduction fuel substitution, and least-cost dose reduction fuel substitution scenarios) achieve the same GWP reduction in our study, the health benefits of these three alternative scenarios are somewhat different (See Tables 6 and 7).
- The Fuel Substitution strategy seems to have more potential to reduce GHG emissions in the long run than the Energy Efficiency strategy, mainly because of a higher degree of benefit in the household sector.
- Because of its extensive use of dirty solid fuels in close proximity to the population on a daily basis, fuel switching and efficiency measures in the household sector have the immediate potential for substantially greater improvements in public health than do such measures in the power sector. Eventually, however, once clean fuels have reached all households, the relative scale of marginal benefits may reverse.
- Combining the Energy Efficiency and Fuel Substitution scenarios together would not produce effects equal to a simple addition of the two because the two scenarios are not completely independent.
- Although GHG reduction is usually accompanied by increasing costs, when the economic benefits of improved health impacts are counted there would be a net economic benefit for the household energy options shifting from traditional coal use (See Figure 17).
- The economic benefits of improved health impacts in the electric power sector, however, are not large enough to offset the incremental economic costs associated with GHG reduction (See Figure 18).
- Exposure-effectiveness (PDE) calculations provide a route to achieving much more cost-effective control of health-damaging pollution than simply using ambient concentrations as indicators of risk.
- GWP and dose (exposure) averted are better indicators of the global warming and health implications of technologies than the raw GHG and HDP emissions.
- Changes in CH_4 GWP to reflect different time horizons do not affect the results significantly. Thus, it is reasonable to use the 20-year GWP of methane in such calculations.

It is likely that other countries with high dependence on solid fuels in the household and power sectors, India for example, could be expected to have similar relationships between GHG reduction with health benefits. Such near-term "secondary" benefits of GHG control provide the opportunity for true "no-regrets" GHG reduction policies in which substantial advantages accrue even if the impact of human-induced climate change itself turns out to be less than many people now fear.

These results may also have important implications for international emissions trading in the form, for example, of joint implementation and clean development mechanisms.

Because the near-term health improvements are local, they accrue nearly entirely to the nation in which GHG-control projects are undertaken. This is unlike the benefits of GHG reductions themselves, which accrue globally. Such large local benefits may provide a significant extra incentive for some developing countries to enter into arrangements by which local GHG controls are financed externally and the GHG emissions credits are shared. Indeed, this study shows that a GHG reduction strategy can actually be consistent with such critical national development objectives as reducing local air pollution, increasing energy efficiency, and improving social equity by providing energy services to remote areas through clean energy sources.

To achieve these benefits, however, considerations of health and other “secondary” benefits need to be included from the start in designing GHG control strategies. Health-based analyses, therefore, deserve a prominent place at the table in international negotiations on GHG control regimes.

Acknowledgments: We are thankful for the support and contributions of John Holdren and Catherine Koshland in Wang (1997), which formed the basis of much of this report. We thank Dilip Ahuja, John Dixon, William Hohenstein, Todd Johnson, Kseniya Lvovsky, Jayant Sathaye, and Robert Taylor for their assistance and appreciate comments on earlier drafts by Robert Chapman, Carlos Corvalan, Devra Davis, Keith Florig, Tord Kjellstrom, Tony McMichael, Rudi Slooff, and Shou-zheng Xue. We thank the former WHO Office of Integrated and Global Environmental Health and the Fogarty International Center for supporting this research.

Appendix A Policy Approaches and Resulting Energy Scenarios

This research examined four energy scenarios, Business as Usual (BAU), Energy Efficiency, Least-cost GWP Reduction Fuel Substitution, and Least-cost Dose Reduction Fuel Substitution Scenarios. The BAU scenarios in the power and household sectors in this research are primarily based on the World Bank GHG scenarios for China (1994b). As explained about the existing GHG energy scenarios in Appendix B, a considerable amount of supplementary assumptions and analysis is required for the three alternative scenarios. The supplementary analysis is based as much as possible on published technology improvements, historical data, international comparisons, and Chinese socio-economic/technology situations.

Changes in the end-use energy intensities are considered in the BAU scenario, and will be held constant among all the scenarios. In reality, however, the higher energy prices resulting from more efficient and cleaner technologies in the alternative scenarios could cause the end-use energy intensities to decline. For simplicity, this research assumes that the end-use energy intensities are constant among all the scenarios.

The Business-As-Usual scenario assumes that the current state of technologies will not change significantly, and that present institutional and policy structures will continue over the 1990-2020 period. The BAU scenario is not a Doing Nothing Case; rather, it considers the current trend of energy efficiency[1] improvement and fuel substitution in both the power and household sectors. It should not be misinterpreted as a prediction of what will actually happen in China.

The Energy Efficiency scenario maintains the same fuel mix as the BAU scenario, but accelerates the improvement in supply-side energy efficiency to achieve the GHG reduction target. In the power sector, the efficiency of power generation technologies is increased at a much faster pace than that in the BAU scenario (See Table 4 in the text section 2.2). The changes in efficiency of power technologies are taken from the World Bank GHG scenarios (1994b). See previous estimates of pollution reduction by improved energy efficiency in China in Li et al. (1995).

In the household sector, Tables A1 and A2 list the assumptions of household cooking and heating for the four scenarios in urban and rural areas respectively by 2020. The BAU scenario for urban household cooking and heating adopts the World Bank projection for China (1994c). For urban heating, raw coal use would shift to coal briquettes, improved coal stoves, central heating, and district heating at a much faster pace than in the BAU. For urban cooking, the Energy Efficiency scenario assumes that all traditional coal use will be replaced by coal briquettes and improved coal stoves by 2000. Under the BAU scenario, however, 5% of the residential households will still use traditional raw coal stoves for cooking from 2000-2020.

[1] The BAU scenario considers improvement in energy efficiencies at both supply-side and demand-side.

Table A1 Assumptions of Household Cooking and Heating for the Four Scenarios in Urban Areas By 2020[1]

Heating	Coal standard[2]	Coal high[2]	Briquette standard[3]	Briquette high[3]	Central heating	District heating	Electric
1990	18%	5%	25%	10%	35%	6%	1%
BAU	5%	5%	17%	8%	40%	15%	10%
Efficiency	0	11%	6%	27%	30%	25%	1%
Substitution	5%	5%	17%	8%	40%	15%	10%
Cooking	Coal standard[2]	Coal High[2]	Briquette standard[3]	Briquette high[3]	Coal gas	Natural gas/LPG	Electric
1990	14%	1%	30%	32%	4%	18%	1%
BAU	5%	5%	25%	39%	5%	20%	1%
Efficiency	0	0	15%	59%	5%	20%	1%
Least-cost GWP	0	0	23%	40%	5%	31%	1%
Least-cost Dose	0	0	15%	42%	5%	30%	8%

Sources: World Bank, 1994c; Wang, 1997; and authors' own estimates.
Notes:
1. The numbers in the table represent the percentage of households that use each energy option by 2020.
2. Raw coal stove standard efficiency and raw coal stove high efficiency
3. Coal briquettes stove standard efficiency and coal briquettes stove high efficiency

The BAU scenario for rural cooking and heating is taken from Wang (1997). For rural heating, traditional biomass use would be phased out more quickly than in the BAU. The Energy Efficiency scenario assumes that all traditional biomass use is replaced by improved biomass stoves by 2000. Traditional coal use would be replaced by coal briquettes and improved coal stoves. In the BAU scenario, traditional coal use is increasing, as a result of decline in biomass use. In the Energy Efficiency scenario, however, traditional raw coal use will be phased out by 2020, replaced by coal briquettes and improved coal stoves. For rural cooking, traditional coal use is replaced by coal briquettes and improved coal stoves more quickly than that in the BAU. Traditional biomass use will be phased out by 2000, replaced by improved biomass stoves.

The Fuel Substitution scenario is intended to explore the pathway of fuel switching to reach the same GHG reduction target. We examined two fuel switching pathways: least-cost per unit GWP reduction scenario and least-cost per unit dose reduction scenario. ("Least-cost" refers to a pathway in which the cheapest options are taken first until exhausted, followed by the next cheapest, etc.) Figures 15 and 16 in the text showed the least-cost curves for household cooking options by 2020.

Table A2 Assumptions of Household Cooking and Heating for the Four Scenarios in Rural Areas By 2020

Heating	Coal standard	Coal High	Briquette standard	Briquette high	Biomass standard[1]	Biomass high[1]		
1990	10%	5%	5%	0	20%	60%		
BAU	20%	10%	15%	5%	5%	45%		
Efficiency	0	0	20%	30%	0	50%		
Least-cost GWP	5%	20%	20%	5%	0	50%		
Least-cost Dose	5%	20%	20%	15%	0	40%		
Cooking	Coal standard	Coal High	Briquette standard	Briquette high	Biomass standard	Biomass high	Biogas	LPG
1990	10%	5%	5%	0	17%	60%	2%	1%
BAU	20%	5%	15%	5%	5%	35%	3%	12%
Efficiency	0	10%	10%	25%	0	40%	3%	12%
Least-cost GWP	5%	12%	10%	10%	0	40%	8%	15%
Least-cost Dose	0	15%	10%	15%	0	35%	10%	15%

Sources: Wang, 1997; and authors' own estimates
Note: 1. Biomass stove standard efficiency and biomass stove high efficiency (i.e. improved biomass stove)

Table A3 lists the assumptions of the energy mix for the BAU and Fuel Substitution scenarios in the power sector by 2020. The BAU scenario is based on the World Bank GHG study (1994d). As seen in the table, conventional coal-fired power plants are replaced by low-carbon options including natural gas, hydro, nuclear, and renewables at a much faster rate than that in the BAU scenario. As demonstrated in Figure 13 in the text, shifting from coal power to natural gas has higher health benefits than GWP reduction. Thus, natural gas plays a more important role in the least-cost dose reduction scenario than in the least-cost GWP reduction scenario.

For urban heating, coal is the primary fuel. There is no fuel substitution for urban heating. That is, the Fuel Substitution scenario for urban heating is the same as the BAU. For urban cooking, coal use is replaced by natural gas/LPG, coal gas, and briquettes at a faster rate than that in the BAU. As shown in Figures 15 and 16, coal gas is the second most expensive cooking option on both the least-cost curves for GWP and dose reduction, thus, the percentage of coal gas in the Fuel Substitution scenarios will remain the same as that in the BAU. The electric stove, on the other hand, has the highest cost per unit GWP reduction, but mid-range cost per unit dose reduction. Hence, the Least-cost GWP Reduction scenario keeps the same percentage of electric stove as the BAU, while the Least-cost Dose Reduction scenario has a much higher percentage of electric stove use than the BAU scenario.

For rural cooking and heating options, biomass use should be first brought down to a renewable supply level, then coal use is replaced by renewable biomass use. For rural cooking, solid fuels (coal and biomass) are replaced by biogas and LPG at a faster rate than those in the BAU. As shown in Figures 15 and 16, improved biomass stoves have the least-cost per unit GWP reduction, but have an increased dose effectiveness compared to coal use. Thus, the Least-cost Dose Reduction scenario has a lower percentage of biomass use than the Least-cost GWP Reduction scenario.

Table A3 Assumptions of Energy Mix for the BAU and Fuel Substitution Scenarios By 2020

Fuel Mix	BAU	Least-cost GWP	Least-cost dose
Coal	75.9%	63.3%	61.9%
Oil	1.1%	1.1%	1.1%
Gas	0.5%	2.5%	3.8%
Hydro	15.9%	19.3%	19.3%
Nuclear	5.3%	10.3%	10.3%
Solar	1.0%	2.6%	2.6%
Wind	0.1%	0.7%	0.7%
Biogas	0	0.1%	0.1%
Geothermal	0	0.1%	0.1%

Sources: World Bank, 1994d; and author’s own estimates.

Appendix B Review of Existing Energy Scenarios

We examined existing GHG reduction scenarios for China, including those of the IPCC, USEPA, LBNL, and the World Bank, but found that none could be directly used in our research.

The IPCC IS92 emission scenarios are constructed by two alternative models: Atmospheric Stabilization Framework (ASF) developed by US EPA and Integrated Model for the Assessment of the Greenhouse Effect (IMAGE) developed by RIVM (IPCC, 1992). ASF was used to estimate future emissions, and the Dutch reviewed the estimates and used IMAGE to validate the results (IPCC, 1992).

In the IPCC IS92 emission scenarios, traditional biomass use is not considered for Centrally Planned (CP) Asia[2] (IPCC, 1992). Traditional biomass use in China, however, was 7.8 EJ in 1995, accounting for about 20% of China's total energy (SETC, 1996). In addition, fuelwood consumption for energy purposes in China was 40% more than its sustainable supply in 1990 (World Bank, 1994e). Thus, CO_2 emissions from deforestation in China was at least 27 million ton carbon, based on carbon emission factor of 29.9 ton/TJ solid biomass fuels (IPCC, 1997). In the IPCC IS92 scenarios, however, CO_2 from deforestation in CP Asia is counted as zero (IPCC, 1992). In addition, the projection of electricity generation capacities in the IPCC IS92a (BAU) scenario is way too conservative and outdated (See table B1).

US EPA updated the IPCC IS92 emission scenarios using the same ASF model for the WRI/WHO joint project of Global Burden of Fossil Fuel on Public Health in 1997. They examined two scenarios: the BAU and policy scenarios, in which CO_2 emissions are reduced to 15% below the 1990 level for Annex I countries and 10% below BAU for non-Annex I countries by 2010 (Lancet, 1997). In the updated EPA policy scenario, only improvement in energy efficiency is considered, and fuel substitution is not counted. The updated EPA/ASF model did not take into account of traditional biomass use and CO_2 emission from deforestation for CP Asia either.

Table B1 compares the coal use in the power and household sectors under the BAU scenario from different existing projections. As shown in table B1, coal use in the electric power sector under the BAU scenario from the updated EPA scenario for CP Asia is about 50% higher than the actual coal use in China in 1990, while the future coal use is lower than the World Bank projection for China. In addition, coal use in the household sector from the updated EPA scenario is 150% higher than the actual 1990 consumption.

The World Bank GHG scenarios for China (1994b) included three scenarios: BAU, Energy Efficiency, and Alternative Energy Scenarios. Under the High Efficiency scenario, the efficiency of power generation technologies is increased at a much faster pace than that in the BAU (World Bank, 1995). Under the Alternative Energy scenario, conventional coal-fired power plants are replaced by low-carbon options including natural gas, hydro, nuclear, and renewables at a faster rate than that in the

[2] CP Asia primarily includes China, Vietnam, Laos, and North Korea. The energy use in China should account for more than 90% of the total energy consumption in CP Asia.

BAU scenario (World Bank, 1994d). For the household sector, however, the World Bank study did not project an alternative scenario. Thus, our own assumptions are necessary to project Energy Efficiency and Fuel Substitution scenarios for the residential sectors.

Table B1 Coal use under BAU from different scenarios (Mt)

	1990	2000	2010	2020	Region
Power					
World Bank	250	439	873	1307	China
IPCC 92	280	378	495	672	CP Asia
EPA/WRI 97	378	611	765	934	CP Asia
Residential					
World Bank	167	169	202	164	China
EPA/WRI 97	425	565	584	565	CP Asia

Sources: IPCC, 1992; the World Bank, 1995; and Lancet, 1997.
Notes: 1. The coal use in the household sector from the World Bank scenario is the sum of coal use in both urban and rural areas.
2. The coal use in the household sector from the EPA scenarios is both residential and commercial coal use for CP Asia[3].

We also examined GHG energy scenarios for China projected by LBNL (Sathaye et al., 1991), East-West Center (EWC)/Tsinghua University (EWC et al., 1994), and Tsinghua University (He et al., 1996). These scenarios either did not provide information of energy use broken down by sectors, or were outdated. Thus, they can not be directly used in our research. In addition, few existing energy scenarios for China projected alternative GHG reduction scenarios for the rural residential sector.

[3] More than 90% of energy use in CP Asia, however, is from China, and commercial energy use is only 10% of residential energy use in China.

Appendix C Economic Assessment

This study employed a levelized cost method to compare the life cycle costs of energy alternatives that deliver the same energy services to users. First, capital costs are calculated using a simple capital recovery factor (CRF) method[4]. This method divides the capital cost into an equal payment series -- an annualized capital cost -- over the lifetime of the equipment. A uniform real discount rate of 12% is chosen[5]. Then the annualized capital costs are added to the annual operation and maintenance costs (O&M) and the fuel costs to obtain the levelized costs. The simple levelized cost method is used here in order to make this research as transparent as possible, while still presenting reasonable estimates of the relative costs of different means to deliver needed energy services.

All costs are in constant 1990 US dollars. Where necessary, RMB Yuan in the year cited is deflated to 1990 values using GDP deflators, and then converted to the US dollars using 1990 annual average exchange rate.

Economic cost of power generation technologies is based on per delivered kWh. First, a levelized cost method is employed to calculate the life-cycle costs for various electricity generation technologies. Then, transmission and distribution (T&D) costs are included where applicable in order to calculate the delivered costs. To deliver 1 kWh of electricity to a user, for example, requires 1.18 kWh of power to be generated when 15% T&D losses are assumed. Therefore, the delivered power costs (cents/kWh delivered) equal the generation costs (cents/kWh generated) plus the T&D costs (cents/kWh generated), and then multiplied by 1.18 kWh (OTA, 1992). Thus, it is the delivered electricity costs from different power technologies, not only the busbar costs, that are compared.

The economic costs of household cooking and heating options are calculated on the basis of per unit of final useful energy using a life-cycle levelized cost method. The capital costs and O&M costs are the economic costs at the point of consumption, while the economic costs at the point of fuel production are included in the fuel costs.

The calculation of levelized costs per unit GJ useful heat supplied for different cooking and heating options is accomplished in six steps: (i) useful energy demand in each household is determined; (ii) annualized capital investments as well as O&M costs of stoves in each household are calculated; (iii) fuel requirement per household for each energy option to supply the same amount of useful energy is determined on the basis of the end-use efficiencies; (iv) the fuel requirements are multiplied by the unit fuel costs[6] to obtain the total fuel costs per household; (v) the levelized costs per household are

[4] The CRF = $\{i(1+i)^n\}/\{(1+i)^n-1\}$ where i is the discount rate and n is the lifetime or period of capital recovery of the systems.

[5] The guidelines by State Planning Commission (SPC) of China for economic evaluation of government projects call for the use of discount rate of 12% per year. In addition, the World Bank's energy projects in China all use 12% per year as discount rate. The sensitivity analysis shows that the choice real discount rate does not affect the final cost results much (Wang, 1997).

[6] Because household fuel prices are heavily subsidized in China, this research used the true market fuel prices, not the subsidized fuel prices, are used as the substitute for marginal costs of fuels.

calculated by adding annualized capital costs, O&M costs, and the fuel costs; (vi) the levelized costs are divided by the useful energy in each household.

Appendix D Dose Effectiveness Calculations

Human inhaled dose is calculated by multiplying exposure concentration, exposed time, affected population, and people's breathing rate. The calculation of particulate dose effectiveness[7] requires the following steps: (a) converting the emissions into equivalent exposure concentrations; (b) converting concentrations into exposures; (c) converting exposures into inhaled dose; and (d) dividing dose by emissions to determine dose effectiveness.

(a) First, it is necessary to convert the emissions from fuel combustion into equivalent exposure concentrations. This research used data from Sinton et al. (1996) for particulate concentrations in Chinese households. The emissions corresponding to the concentrations over the same period of time need to be determined. Unfortunately, the database (Sinton et al., 1996) did not provide fuel use data corresponding to the indoor concentration owing to a lack of documentation in the original measurement reports. For a first approximation, this study used the average fuel use per household per year for cooking and heating, and the particulate emission factor, to roughly estimate the particulate emissions.

In addition to the emissions from indoor fuel combustion sources, emissions from outdoors also can penetrate indoors, and thus, increase indoor concentration. We are lacking the quantitative information, however, to estimate the emission penetration from outdoors. Thus, the low end of measured indoor particulate concentrations are chosen, and directly linked to emissions from indoor fuel combustion sources.

For coal-fired power plants, we used a Gaussian plume model averaged over the year, as shown in the formula (below), with Chinese meteorology data to estimate the changes in particulate concentration resulting from marginal changes in emissions (similar to the approach in World Bank, 1994a). A stack height plus plume rise of 75 meters is chosen, based on typical stack heights for power plants in China (World Bank, 1994a). The ambient concentrations were estimated for concentric circular bands around the emission source. The bands were centered on 100, 200, through 1,000 meters, and then up to 50,000 meters, with 100 meters as an interval. Since sixteen wind directions are assumed, one sixteenth of the area encompassed by the concentric circles is affected by the emissions at any one time.

The World Bank study (1994a) chose 10 kilometers from the emission source as the boundary. Although the ambient concentrations from per ton particulate emissions decline to a very low level at the 10 kilometers boundary, the actual particulate exposure is not very low. To account for the exposure outside the 10 kilometers boundary, we developed an exposure-distance curve (See Figure 6) to extrapolate the exposure outside the 10 kilometers boundary, as explained in the following section.

[7] Particulate dose effectiveness is defined as gram of particulates inhaled by human per ton particulate emission (Smith, 1987).

The Gaussian plume model formula is as following (World Bank, 1994a):

$$C = (2 / \pi)^{1/2} * Q / (U * X * (\pi / 8) * \sigma_z) * EXP (-1/2 * ((h + \Delta h) / \sigma_z)^2)$$

C	annual ground-level particulate concentration at distance X from stack (grams/m^3)
Q	particulate emissions from the source (grams/second)
U	average wind speed at emission source (meters/second) = 2.6 m/s in Beijing
X	the distance between the emission source and the monitoring site (meters)
σ_z	standard deviation of vertical dilution = $0.44 * X^{0.94}$
$h + \Delta h$	the height of the stack plus plume rise above the stack (meters) = 75 m

To account for the effects of secondary pollutants such as sulfates in the fine particulate (Hidy, 1994), we incorporate estimated SO_2 emissions. To generate 1 kWh electricity, about 8.4 g TSP and 10.4 g SO_2 are emitted[8]. Thus, one ton of TSP emission is accompanied by 1.2 tons of SO_2 emission. We assume that about 50% of this amount of SO_2 emission is converted to sulfur compounds in the fine particulate fraction.

(b) One of the principles of exposure assessment is to measure where the people are, that is, to focus on human exposures rather than ambient concentrations (Smith, 1993). For emissions from coal-fired power plants, the exposed population was estimated by multiplying population density in average Chinese cities[9] (World Bank, 1994a) by the area of concentric circular bands around the emissions source. Then, the exposure is the sum of the product of the particulate concentration and the affected population within each circular band with a 100 meters as an interval, as explained above.

Figure 6 in the text shows the distribution of exposure over the distance from the emission source. As shown in Figure 6, the maximum particulate exposure occurs about 300 meters from the emission source. After that distance, the particulate exposure will gradually decline over distance. At 10 kilometers from the emission source, human exposure reduces to 0.5 ug/m^3-person/ton-year emission. We extended the boundary to 50 kilometers, and extrapolated the exposure from 10 to 50 kilometers[10]. The low exposure level predicted at 50 kilometers (0.1 ug/m^3-person/ton-year) provides the argument to choose this distance as the boundary. Thus, accounting for the exposure from 100 meters extending to 50,000 meters is about 35% higher than accounting for the exposure levels within 10,000 meters alone.

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[8] TSP emission = ash content (25%) x percentage of ash caught in flue gas (70%) x (1 - removal rate (90%)) x heat rate (10,286 Btu/kWh) / fuel heating value (21.42 GJ/ton); SO_2 emission = sulfur content (1.2%) x molecular weight conversion from S to SO_2 (2) x release rate (90%) x heat rate (10,286 Btu/kWh) / fuel heating value (21.42 GJ/ton) (World Bank, 1991).

[9] In the World Bank study (1994a), the population density was derived from the average population density in fifteen Chinese cities (1249 people/km^2), ranging from 700 people/km^2 in Benxi, Liaoning to 5200 people/km^2 in Xuzhou, Jiangsu. In addition, the average population density in the municipalities, which includes both the city and suburb, is estimated at 508 people/km^2.

[10] When we extrapolated the exposure from 10-50 kilometers, we used the same population density as that within the 10 kilometers. This, however, may overestimate the exposure, because the population density may decline at the outskirts of the city. If the actual density were half, the reduction in total exposure would be 0.5 x 0.35 (fraction of exposure beyond 10 km) = 17%

For emissions from household stoves, it is assumed that four people in one family are exposed to the measured indoor particulate concentrations[11].

Next, the exposed time needs to be estimated to determine both indoor and outdoor exposures from power plants and household stove emissions respectively. The outdoor particulate concentration from the coal-fired power plants reach people both outdoors and indoors. It is assumed that urban people spend 90% of the time indoors and 10% of the time outdoors in China. When people stay outdoors, they inhale 100% of the outdoor concentration. It is assumed that indoor concentration from power plants equals 80% of local outdoor concentration (See Figure 7 in the text). Therefore, the average concentration experienced by the population at any one place due to a power plant is about 82% of the calculated outdoor concentration at that point due to the power plant[12].

Similarly, indoor concentrations from household fuel combustion affect people both indoors and outdoors. In China, about 20% of people are children (under 5 years old) and elderly (above 60 years old), who spend 90% of the time at home (Murray & Lopez, 1996). The other 80% of people are labor force and students, who spend 50% of the time in households and the other 40% of the time in office or school. Thus, people spend 58% of time at home[13]. When people stay at home, they inhale 100% of the indoor concentrations from household stove combustion. When people spend 10% of the time outdoors, it is assumed that outdoor concentration from household stoves equals to 10% of indoor level. In addition to 58% of time spent at home and 10% of time spent outdoors, we assume that the other 32% of time spent in indoor places is not affected by household stoves (See Figure 8 in the text). Therefore, the mean exposure concentration experienced by people is about 59% of the indoor particulate concentrations produced from household stoves[14]

(c.) Inhaled dose is the product of exposure and human breathing rates. The population breathing rate, 15.5 m^3/person-d, is calculated as the average breathing rate of one adult man, woman, children, and infant (Smith, 1987).

(d) Then, the particulate dose is divided by particulate emissions to obtain grams inhaled by humans per ton of particulates emitted.

Household stove coal smoke is taken to illustrate the calculation of particulate dose commitment:

PM_{10} emissions = (2 kg/h) x (15 kg/ton) x (2.8 h/d) x (365 d/yr.) = 30 kg/yr.
(Fuel use 2 kg/h; TSP emission factor 20 kg/ton, from World Bank, 1991; PM_{10} = 75% TSP; cooking time 2.8 h/d);

[11] The indoor particulate concentrations are measured during 12-24 hour period (Sinton et al., 1996).
[12] 10% x 100% (outdoor exposure) + 90% x 80% (indoor exposure) = 82%
[13] 20% (of population) x 90% (of time at home) + 80% (of population) x 50% (of time at home) = 58%
[14] 58% x 100% (indoor exposure) + 10% x 10% (outdoor exposure) = 59%

Inhaled dose = (150 ug/m^3) x (4 persons) x (59%) x (15.5 m^3/d) x (365 d/yr.) = 2 g/yr. (Concentration from Sinton et al. 1996: PM_{10} from coal, 24 hours, average for summer and winter in four provinces, Qin et al., 1990; breathing rate from Smith, 1987)

Dose commitment = (2 g/yr.) / (30 kg/yr.) = 67 g/ton.

Appendix E Exposure-Response Relationships for HDP

Existing studies provided a wide range of exposure-response coefficients. In order to capture both acute and chronic effects, some epidemiological studies are conducted on a cross-sectional basis, with different populations receiving different pollution levels over a number of years. In cross-sectional studies, personal factors, such as smoking and diet, are potential confounders because they may correlate both with health and long-term pollution (Smith, 1994b). Because such studies are expensive, only a few have been completed to date, (Dockery et al., 1993 and Pope et al., 1995). The results indicate an average 3% increase in mortality associated with each 10 ug/m^3 increase in annual PM_{10} (Pope et al., 1996).

More common are time-series studies, which correlate daily variation in air pollution with variation in daily mortality in a given city (Cropper et al., 1996). Such studies are not sensitive to individual behavior confounders, but can be sensitive to confounding factors that may affect both pollution and health in the short-term, such as weather (Ostro, 1994). A number of recent US time-series studies on the health impacts of particulate air pollution showed remarkably consistent results in the association between particulate pollution and mortality (Dockery et al., 1992; Pope et al., 1992; and Schwartz, 1991). These results suggest that a 10 ug/m^3 increase in daily PM_{10} is associated with an increase in daily mortality equal to 1%. Recent analyses of 12 European studies show a similar range, although also showing an independent effect of SO_2 (Katsouyanni et al., 1997).

Xu et al. (1994) also conducted time-series studies in Chinese cities. In Beijing, total daily mortality was estimated to increase by 4% with each doubling in TSP concentration (Xu et al., 1994). In Shenyang, a 100 ug/m^3 increase in daily TSP concentration was associated with an increase in daily mortality by 2% (World Bank, 1997). Although fine particulates (PM_{10} or $PM_{2.5}$) is apparently the best measure of health impacts, only total suspended particulate (TSP) have been measured consistently in China. Where needed, we have assumed that PM_{10} is 55% of TSP (Dockery et al., 1996). Within a few years, however, more data specific to China should be available.

In general outline, we have applied the range of risk estimates spanning from those derived coming from short-term time-series studies (lower end) to those coming from long-term studies (upper end) as suggested by Ostro (1996). Although we have incorporated Chinese and other studies not considered in Ostro (1996), we recognize that the two types of studies are not strictly comparable because, among other factors, the degree of prematurity (lost life years) of the deaths determined by each may be quite different. (McMichael et al, 1998).

There are other problems involved in extrapolating risk estimates derived basically by developed-country studies to developing-country conditions, such as pertain in China:

--Differences in pollutant mix due to different sources, i.e. although particulates can be used as indicator of hazard in both cases, biomass fuels produce relatively more organic compounds and fossil fuels more sulfur oxides. Relatively little of pollution mix in many of the US and European studies came from coal, unlike China. Thus risk (exposure-response) estimates derived in the latter situation may not apply to the former.

Differences in exposure patterns, i.e., indoor concentrations tend to vary much more during the day (because of household cooking and heating schedules) than do outdoor urban levels.

Different exposure levels, i.e., the average exposure levels of concern in households using unvented solid-fuel stoves are 10-50 times greater than the levels studied in most recent urban outdoor studies. As has been shown with cigarette smoking, there is likely to be a diminishing of the effect per unit increase in exposure at these high levels.

Different life-exposure patterns. In China, for example, many adults will have been raised in households using dirty solid fuels for cooking and heating, thereby perhaps changing their lifelong sensitivity to airborne pollutants, including outdoor air pollution and active/passive smoking.

Different populations, i.e., the pattern of disease and competing risk factors differ dramatically between urban developed-country populations, the world's richest and most healthy populations, and relatively poor rural people exposed to indoor air pollution in China.

There are also fundamental difficulties in using the results from short-term time-series studies to predict impacts from long-term exposures (McMichael et al., 1998). It would seem that they are most likely to underestimate the full impacts. On the other hand, the available long-term cohort studies may overestimate impacts per unit exposure because current ill-health is due to cumulative exposures over time and, in the European and North American cities where such studies have been done, ambient levels have tended to decrease over time. In China, however, where exposures may not have been decreasing and may have even increased for some populations in recent decades, the situation may be different.

As we state in Section 5, therefore, there is clearly a need for more well-done studies in situations where exposures and health effects are greatest, i.e., the cities and solid-fuel using households of developing countries such as China. Until such studies are done, we must rely on estimates with uncomfortably large ranges to take into account the many uncertainties of extrapolation.

Appendix F Economic Benefits of Reduced Health Impacts

Based on the reduced health impacts, this research also estimated the marginal economic benefits of the avoided mortality and morbidity associated with moving away from conventional coal use in the power and household sectors. There is no universal or universally accepted way to convert improved health impacts into economic benefits but rather several possible approaches. This study adopted the World Bank estimates (1994a).

The estimate of China's willingness to pay for risk reduction in premature death in the World Bank study (1994a) first followed Ostro's method, that is, the value to prevent one statistical death in developing countries is worth 24,000 times daily wages, based on the assumption of $3 million per statistical life saved in the US. Then, the purchasing power parity (PPP) ratio is used to adjust for the differences in the levels of purchasing abilities between wages earned in the US and China. As a result, the value of a statistical life in China is estimated at $123,700, about one twenty-fourth the US value.

The economic evaluation of avoided morbidity has been less successful than that aimed at avoided mortality (Cropper et al., 1992). The World Bank study (1994a) used wages directly as an estimate of the value of work days lost due to illness. The economic valuation of a restricted activity day adopts wages as a basis (0.464 x the wage), and then is further adjusted by the PPP ratio (World Bank, 1994a). A hospital admission is assumed to require 15.9 days of lost work and cost 29.2 Yuan in medical expenses per day (World Bank, 1994a). As a result, the economic values of a restricted activity day and a hospital admission are estimated at $2.4 and $124.4 respectively. Use of wages, however, does not account for the avoided physical and mental discomfort, thus, these estimates are conservative.

Table F1 lists the economic benefits of improved health impacts resulting from one ton reduction in particulate emission from the a power plant and a household stove. The economic values are calculated by multiplying the central estimates of averted mortality and morbidity resulting from one ton reduction in particulate emission by the economic value of preventing a premature death and a morbidity outcome. As shown in Table F1, preventing one ton in particulate emission from a household stove has much higher economic benefits than that from a power plant. Therefore, investing in air pollution control in the household sector is much more cost-effective to improve human health impacts than investing in the power sector.

The marginal economic benefits of reduced health effects associated with moving away from conventional coal use is calculated by three steps: (a) converting the changes in emissions to the changes in exposures; (b) converting the changes in exposures to averted mortality and morbidity outcomes; and then (c) economically valuing the mortality and morbidity. When the economic benefits of improved health impacts are taken into account, the net economic costs of GHG reduction is determined by deducting the economic benefits from the incremental costs of the changes in energy systems.

Table F1 Economic Benefits of Improved Health Impacts from One Ton Reduction in Particulate (US$/ton)

	Power	Household
Mortality	245	10,000
Morbidity		
Hospital admission	0.4	18
Restricted activity day	25	1640

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GLOSSARY

BAU	Business as Usual. Scenario assuming no special effort to control greenhouse gases or health-damaging pollutants beyond what is now planned.
Dose	measured as milligrams (mg) indicating how much particulate pollution is actually breathed in by a population.
Dose commitment or effectiveness	the fraction of material emitted from an air pollution source that is actually inhaled by people, g/t or mg/kg.
EJ	exajoule (10^{18} joule). Energy unit = 10^{9} GJ
Energy Efficiency Scenario	Energy Efficiency Scenario maintains the same fuel mix as the Business As Usual scenario, but accelerates the improvement in supply-side energy efficiency to achieve the GHG reduction target.
Exposure Unit	See (ug/m3)-person-y
Fuel Substitution Scenario	Fuel Substitution Scenario is intended to explore the pathways of fuel switching to reach the same GHG reduction target as the Energy Efficiency Scenario, while maintaining t he same energy efficiency as the Business As Usual scenario.
GDP	Gross Domestic Product
GHG	Greenhouse Gas
GHG Reduction Target	10% below Business As Usual by 2010 and 15% below Business As Usual (BAU) by 2020 for China
GJ	gigajoule (10^{9} joule). Energy unit = 1000 MJ
GWP	Global Warming Potential, in equivalent kg carbon as carbon dioxide per kg carbon in non-CO_2 greenhouse gases.
HDP	Health-Damaging Pollutant
IMF	International Monetary Fund
IPCC	Intergovernmental Panel on Climate Change

joule	Energy unit (J) = 0.24 calories
kgce	kilogram (10^3 gram) coal equivalent. Energy unit = 30 megajoule (MJ)
kgC	kilograms carbon. Way of combining carbon-containing gases, such as carbon dioxide and methane
kgC-CO_2	kilograms carbon as carbon dioxide. Global warming unit = sum of kgC of each GHG times its GWP. Here, the GWP of methane is taken as 25 over a 20-year period. That is, one mole of methane has the global warming effects of 25 moles of carbon dioxide over a 20-year period. By definition, the GWP of carbon dioxide is 1.0.
kWh	kilowatt-hour (10^3 watt-hour). Energy unit usually applied to electricity = 3.6 MJ = 0.0036 GJ
LBNL	Lawrence Berkeley National Laboratory
Least-cost-per-unit-dose-reduction Scenario	This is one of the two Fuel Substitution Scenarios. This scenario is designed such that a mix of technologies can be chosen to substitute for conventional coal use to reach the GHG reduction target by ranking their co sts per unit dose reduction until the energy demand is met.
Least-cost-per-unit-GWP-reduction Scenario	This is the other Fuel Substitution Scenario. This scenario is designed such that a mix of technologies can be chosen to substitute for conventional coal use to reach the GHG reduction target by ranking their costs unit GWP reduction until the energy demand is met.
LPG	Liquefied Petroleum Gas (bottled gas)
mg	milligram (10^{-3} gram). See Dose.
MJ	megajoule (10^6 joule). Energy unit = 0.033 kgce
MtC	megaton (10^6 tons) carbon. Global warming unit = 10^9 kgC
Mtce	megaton (10^6 tons) coal equivalent. Energy unit.
MWh	megawatt-hour (10^3 kWh). Energy unit usually applied to electricity.

Near-term health benefits	reduction in the chronic and acute impacts of air pollution in the first two decades of the 21st century.
NMHC	Non-Methane Hydrocarbon
PDE	Particulate Dose Effectiveness (see dose effectiveness, above)
PPP	Purchasing Power Parity, a means of adjusting different currencies according to their actual ability to purchase a common set of goods and services, i.e. corrected for local prices.
SETC	State Economic & Trade Commission, China.
SPC	State Planning Commission, China.
T&D	Transmission & Distribution (of electricity)
(ug/m3)-person-y	Exposure Unit indicating the pollutant concentration that has been experienced by how many people for how long.
TJ	terajoule (10^{12} joule). Energy unit = 1000 GJ
TWh	terawatt-hour (10^{9} kWh). Energy unit usually applied to electricity.
WTP	Willingness To Pay