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Co-benefits of climate policy—lessons learned from a study in Shanxi, China

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Abstract

We analyse health benefits and socio-economic costs of CO_2-abating options related to coal consumption in Shanxi, China. Emission reductions are estimated for SO_2 and particles in addition to CO_2. Co-benefits of each option are estimated in terms of how effectively it improves local air quality and thereby reduces health damage from pollution. The population-weighted exposure level for particles and SO_2 is estimated using air quality monitoring data, and a simplified methodology is applied to estimate the reduced exposure to the population that may result from implementing the abatement measures. Exposure–response functions from Chinese and international epidemiological studies are used to estimate the ensuing health effects. A method for estimating the impact of chronic PM_{10} exposure on life expectancy in the affected population is developed and applied. An economic evaluation of the reduced health effect is made by determining unit prices of health impacts based partly on damage costs and partly on the willingness-to-pay approach. Our assessment of CO_2-reducing abatement options in Shanxi demonstrates that these measures entail large co-benefits and are highly profitable in a socio-economic sense.

Keywords: Co-benefits; Health effects; Clean coal technologies; Air pollution; Climate policy

1. Introduction

The progress in coping with emissions of greenhouse gasses (GHGs) is slow, and initiatives that may speed up the process are necessary. Greater emphasis on the fact that measures primarily intended to reduce GHG often have other benefits, usually denoted co- or ancillary benefits, may be important in this connection. Several studies have shown large co-benefits related to improved local air quality (Aaheim et al., 1999; Aunan et al., 1998; Cifuentes et al., 2001; Lee Davis et al., 1997; Dessus and O'Connor, 1999; Ekins, 1996; Seip et al., 2001; Wang and Smith, 1999a, b). The most important co-benefits are reduced damage to human health and reduced corrosion rates of materials. On a more regional scale, reduced emission of air pollutants may also contribute to reducing crop losses that are brought about by surface ozone and regional haze.

Generally, the largest local and regional air pollution problems are found in countries that do not have emission-reduction obligations in the Kyoto Protocol. Human exposure to air pollution is particularly acute in megacities in many developing countries, often due to a heavy reliance on coal in power production and outdated technology in automobiles. Because typical 'first generation' air pollution abatement measures, such as removing particles and SO_2 from the waste gases, often have not been implemented, the co-benefits of typical GHG abatement measures may be large in developing countries. While the climate change issue is usually not high on the political agenda, reducing urban air pollution often is. Since progress in the efforts to mitigate GHG emissions to a large extent depends on involvement of developing countries, the recognition of large co-benefits may have an important impact on future international negotiations on the Climate Convention.

A possibility for speeding up implementation of energy-efficient technologies in developing countries has emerged with the *Clean Development Mechanism*

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(CDM), which is one of the three flexibility mechanisms in the Kyoto Protocol. Through the CDM, developed countries (Annex I countries to the Framework Convention) may invest in GHG mitigation projects in developing countries (non-Annex I countries) in order to be credited for the reductions. This emerging financing option now seems more available after the negotiations in Marrakech in 2001 about the implementation of the Protocol, although the withdrawal of the USA may affect the compliance costs of the remaining countries in a way that does not favour the CDM (Löschel and Zhang, 2002). However, since the prospects that enough countries will ratify the Protocol so that it could enter into force are perhaps more promising than ever, it should be expected that industry in Annex I countries will show increased attention to investments in CDM projects.

For developing countries, the CDM gives an economic incentive to reduce their GHG emissions and provides an opportunity to take an active part in the global process of slowing down emission growth. However, economic incentives may not be sufficient. Political priorities and transaction costs reduce the practical scope for GHG mitigation measures under the CDM. In this situation, awareness of the co-benefits may contribute to GHG mitigation in a way that is constructive both to developing countries (seeking to improve the local and regional environment) and to companies in developed countries (seeking to demonstrate 'Corporate Social Responsibility' as well as gaining GHG credits).

As a country with large GHG emissions and many areas with high local pollution levels, China is of particular interest. With about 14% of the world's emissions of carbon dioxide, China now is the world's second largest CO_2 emitter. In the last decade, it was the largest producer and consumer of coal (IEA, 1998). In 1998, coal generated 72% of China's energy (National Bureau of Statistics, 1999a). According to 'Business as Usual' projections by the IEA (1998), the country is expected to continue to rely on coal and to surpass the present number-one CO_2 emitter, the US, by 2020. Chinese CO_2 emissions are expected to grow despite considerable success in reducing the CO_2 intensities (Garbaccio et al., 1999; Zhang, 2000).

The purpose of this paper, which describes the results of a case study in Shanxi, the main coal-producing province in China, is to indicate the magnitude, in physical and economic terms, of the local and regional benefits that implementation of GHG mitigation measures may provide host countries. We aim at identifying options with large co-benefits, primarily in terms of reduced damage to human health. Some qualitative statements regarding possible positive effects on agricultural production and building materials are also included. In the first phase, documented in this paper, we have focused on analysing the environmental impacts of six different abatement options that are mainly applicable in the industry sector, the power sector and in rural households. The results are primarily intended to illustrate the importance of considering local, regional and global environmental problems in an integrated way. More details on some of the issues are found in Aunan et al. (2000a).

2. The Shanxi province

Shanxi lies in the central parts of North China covering an area of 156,300 km^2 and is bordered by the Yellow River (Huang He) in the west. The province is to a large extent mountainous and has a continental monsoon climate, which means that most of the precipitation falls in summer. The province contains 118 counties with its capital set in Taiyuan City. The total population of Shanxi is 31.7 million (1998), of this we estimate that about 16.2 million live in urban areas. Generally, the central axis of the province has the highest population density, the most populated counties being Taiyuan and Datong. Shanxi has 2.13 million hectares of forest, which cover 13.8% of the province's land surface area (figures for 1990).

Shanxi is one of China's major energy bases with rich coal and iron deposits. The coal industry is one of the most important industries in the province (coal and coal-related industries accounted for about one-third of the total output from industry and agriculture in 2000 (Zhang, 2002)), and coal production in Shanxi represents about a quarter of total production in China. Shanxi has eight major mining areas and 3000 or more medium-sized and small mines (CERNET, 2000). About two-thirds of the production is exported as raw coal. Coke-making plants process more than 50% of the coal retained. The coke production in Shanxi represents nearly half the coke produced in China. About two-thirds of the coke produced in Shanxi is normally exported to other provinces and countries. The consumption of coke within the province takes place to a large extent in the metallurgical industry.

Heavy industry in Shanxi includes production of machinery, e.g. tractors, locomotives, automobiles and equipment used in mines and the metallurgical industry. Main light industry sectors are textile, paper and food. As compared to some of the coastal provinces, which have benefited from a favourable economic policy and substantial economic growth after the 1978 reform, Shanxi has, together with other central and western provinces, been lagging behind. In 1998, the GDP/capita in Shanxi was 79% of the average in China (National Bureau of Statistics, 1999a).

3. Air pollution in Shanxi cities

The main sources of air pollution in major Shanxi cities are coal mining, coking, power plants and metallurgical industries. In addition comes the widespread use of coal in small coal-fired commercial boilers for heating and steam generation as well as for heating and cooking in the household sector.

The second column in Table 1 shows the annual coal consumption in Shanxi (average for the period 1995–1997). The largest single consumer is the coke-making sector. Using these figures, we have estimated the total emissions of total suspended particulates (TSP) and SO_2 from coal consumption in Shanxi (Table 1, last two columns). Emission factors for TSP per ton of coal range from 2.9 kg for the coke-making industry to 20 kg for households. For SO_2 the values range from 5.0 (coke making) to 20.2 kg (households). The low SO_2 emission factor for coke making reflects both the generally lower sulphur content in coal used for coke making and that much of the coke is exported from the province (some sulphur is retained in the coke). Details are given in Table 8 in the appendix. Our estimate of total SO_2 emission is somewhat higher than that given by National Bureau of Statistics (1999a).

Table 1
Annual coal consumption (million tons of raw coal) and estimated related emissions of TSP and SO_2 in Shanxi, average for 1995–1997

	Consumption (million tons raw coal)	% total	TSP (1000 tons)	SO_2 (1000 tons)
Total	136.15	100.0	667.8	1532.7[a]
Households	5.88	4.3	117.8	119.0
Rural	0.97	0.7	19.4	19.6
Urban	4.92	3.6	98.4	99.4
Industry	130.26	95.7	550.0	1413.7
Power	28.33	20.8	102.0	512.8
Coke making	72.06	52.9	209.0	360.3
Industrial combustion	29.87	21.9	239.0	540.6

[a] This figure is somewhat higher than other estimates, e.g. by the National Bureau of Statistics (1999a, b) and Streets and Waldhoff (2000).

The air pollution situation is severe in many cities of Shanxi, as indicated by the fact that the levels of SO_2 far exceed World Health Organisation (WHO) guidelines, which is $50\,\mu g/m^3$ as an annual average (WHO, 1995), see Fig. 1. The WHO has recommended that no guideline should be set for particulate matter because there is no evident threshold for effects on morbidity and mortality. In the absence of WHO guidelines, the PM_{10} air quality standard of $40\,\mu g/m^3$ (annual average) given in the EU Council Directive 99/30 may be used to assess the severity of the air pollution levels (EU, 2000). Assuming a PM_{10}/TSP conversion factor of about 0.55 (see Aunan et al., 2000a), Fig. 1 shows that this threshold is exceeded many times over in all cities. Due to large variations in the concentration level during the year, with high maximum levels during the winter in many cities, the situation may be even worse than the annual average values indicated. The NO_x levels are fairly low, and taking into consideration that NO_2 constitutes only a share of NO_x, the WHO guideline of $40\,\mu g/m^3$ (annual average NO_2 level) is probably not exceeded in most cities.

The province capital, *Taiyuan*, is situated in a mountain basin in central Shanxi, surrounded by hills and mountains on three sides. The topography leads to

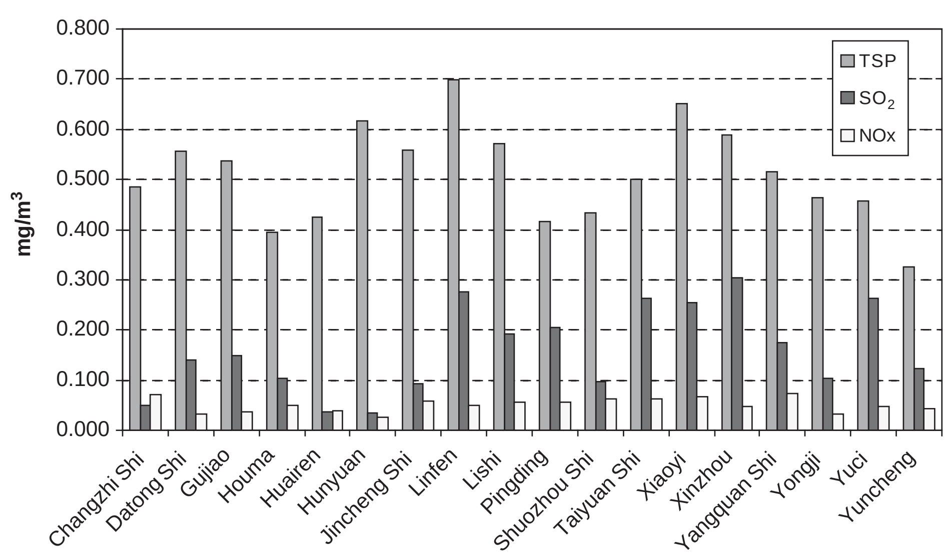

Fig. 1. Air quality in some cities in Shanxi (average for 1997 and 1998). *Source*: Shanxi EPB, 1999.

periods of inversion and stagnant air masses during the winter, thus enhancing the concentration of air pollutants. The main emission sources in Taiyuan are two power plants, with installed capacities of 1000 and 600 MW, respectively, and the Taiyuan Iron and Steel Company, which is one of the major iron and steel complexes in China. In Taiyuan these three sources represent about 75% and 67%, respectively, of the total non-residential SO_2 and TSP emissions, and are all situated in central areas of the city. Taiyuan also has a substantial cement production. The main emissions from iron and steel production companies relate to coke production (for own use), power production for own electricity supply, and the process emissions (blast furnace). Besides the Taiyuan Iron and Steel Company plant, there are medium-sized and small iron and steel works in Changzhi, Linfen and Yangquan. *Datong* is the second largest city in Shanxi and the economic centre of northern Shanxi. Northern Shanxi is one of the main coal mining regions in China, and mining and coking are major air pollution sources in the area. The city of Datong also has machine-building and building-materials industries, as well as a substantial cement production.

How many are exposed to the rather severe concentrations of air pollutants? Based on air quality data from Shanxi cities (Shanxi EPB, 1999), an annual average population-weighted exposure (PWE) can be calculated from

$$\mathrm{PWE} = \frac{1}{P}\sum_i c_i P_i,$$

where P_i is the urban population in city i, c_i is the average level of the pollutant in city i, and P is the total urban population in all cities (we used data for the so-called 'non-agricultural population' as a proxy for the number of urban residents in the cities). Again assuming $PM_{10} = 0.55$ TSP, the PWE for PM_{10} in the 18 cities for which we had data is 280 $\mu g/m^3$; the corresponding value for SO_2 is 186 $\mu g/m^3$.

4. Abatement measures

Six different abatement options that will reduce the emissions related to use of coal were analysed with respect to cost and emission-reduction potential (CO_2, PM_{10} and SO_2). The options and the amount and share of the present coal consumption that may feasibly be regulated by each option are listed in Table 2. The costs and emission-reduction potential are listed in Table 3. We have looked at options feasible for the industry sector, the power sector and rural households:

- 'Coal washing' removes coal dust as well as impurities in the coal. Washed coal in Shanxi is roughly 30% more expensive than unwashed coal (US$5 per ton coal). The cost lies in the operation, the water, plus the loss of some useful coal. The benefit is more efficient operation and approximately 10% lower CO_2 emissions. Moreover, coal washing also greatly reduces particle and SO_2 emissions. The price of water in Shanxi probably does not reflect the underlying scarcity of water. The cost estimate in

Table 2
Potentials for the abatement options

	Rural households	Power	Industrial combustion	Emission reduction potential (%)		
				CO_2	TSP	SO_2
Coal washing (million tons)		23.8	29.9	10	35	40
% of source		84.1	100.0			
Briquetting (million tons)	1.0		29.9	10	35	35[a]
% of source	100.0		100.0			
Improved management (million tons)			22.4	8	8	8
% of source			75.0			
Boiler replacement (million tons)			22.4	25	25	25
% of source			75.0			
Co-generation (million tons)[b]			0.7	21	21	21
% of source			2.3			
Modified boiler design (million tons)[c]		17.0	17.9	17	17	17
% of source		60.0	60.0			

Amount (million tons of raw coal consumed), percentage share of source consumption that may feasibly be regulated by the selected abatement options, and emission reduction potential (as percentage of baseline emission coefficient).

[a] This reduction in SO_2 emission is possible with proper lime addition. Briquettes commonly used in Shanxi at present have much less lime and the reduction in SO_2 emissions is only about 5%.

[b] Paper and textile industries only.

[c] Multilayer combustion system.

Table 3
Costs per ton CO_2, and emission reduction potential for the abatement options

	Abatement cost (US$/ton CO_2)	CO_2 (million tons)	TSP (1000 tons)	SO_2 (1000 tons)
Co-generation	−30.00	0.3	1.2	2.6
Modified boiler design	−6.23	12.8	34.1	105.3
Boiler replacement	−2.74	12.3	44.8	101.4
Improved boiler management	9.20	3.7	13.4	30.4
Coal washing	22.73	11.8	113.6	388.7
Briquetting	27.27	6.8	90.4	196.0

Table 3 therefore probably is a lower bound on the true cost in Shanxi.

- 'Briquetting' binds the coal together. Similar to washing, the effects are more efficient operation and coal dust is eliminated. The cost (approximately US$6 per ton coal) is related to the process and the lime that is added to the briquettes. The reduction in SO_2 emissions is due to the lime. If the full amount of lime is not added, which sometimes happens for cost reasons, the SO_2 reduction, as well as the cost, will be lower.
- 'Improved boiler management' includes simple changes in management practice, maintenance, etc., and small investments. The cost of improvements goes from negative and up. Our cost estimate and potential for improvements are adapted to Shanxi conditions from Fang et al. (1999).
- 'Boiler replacement' amounts to exchanging old, inefficient industrial boilers with state-of-the-art boilers. Still, the new boilers we have looked at are not extravagant, but medium sized, indigenous boilers. A 4 t/h boiler of 75% efficiency can be purchased in China for approximately US$23 000. Assuming 8% depreciation and 8% interest, the annual user cost is US$3650. Assuming that an existing boiler has an efficiency of 60% (Fang et al., 1999), the annual fuel savings amounts to US$4450, i.e. more than the annualised cost. It would have been more still had not the price of coal been low in Shanxi (around US$15 per ton).
- 'Co-generation' of heat and electricity is an extremely low-cost option when the conditions are right, i.e. no infrastructure investments are needed to utilise the heat. These conditions are met in plants in the paper and textile industries. They may be met elsewhere as well, but we do not want to exaggerate the potential. To assess the cost of co-generation in these industries, we use an average informed by two estimates from London et al. (1998), and an estimate from Aarhus et al. (1999). The potential for improvement is from Aarhus et al. (1999).
- 'Modified boiler design' amounts, in this study, to a multilayer combustion system. The purpose of such a system is to sort the coal that goes into the boiler. This allows more efficient operation of the boiler. The multilayer combustion system is a potential option in chain stoker boilers (60% of Shanxi boilers).

Total coal consumption in Shanxi is about 136 million tons (average for 1995–1997, see Table 1). The share of the coal consumption in the different source groups that may feasibly be regulated by the abatement options is assumed to be between 60% and 100%, except for the option 'co-generation' (Table 2) (see also Fang et al., 1999).

5. Exposure–response functions for health effects of air pollution

Several Chinese epidemiological studies indicate exposure–response functions for the association between air pollutants and health effects. In the following, we have applied some of these functions supplemented by results from some studies in Europe and the United States when necessary (Aunan et al., 2000a). Compared to studies in Europe and the United States, the Chinese studies generally report lower coefficients for the exposure–response relationships between air pollution and health effects. Problems related to possible confounding with indoor air pollution are indicated in most of the Chinese studies. We therefore regard the functions based on Chinese studies as rather conservative, i.e. they may possibly understate the effect of air pollution, and subsequently of air pollution abatement. Murray et al. (2001) have considered the various factors that may lead to differences between developed and developing countries in human tolerance of air pollutants, and suggest that a similar range of sensitivity can be expected. Transferring risk estimates from one population to another is, however, encumbered with uncertainties. For instance, the composition of the car fleet in Western Europe and the United States, where most of the epidemiological studies have been performed, differs substantially from that in China. This, together with other differences, such as the widespread use of coal in China, implies that the air pollution mixture (co-pollutants) differs substantially between China and Western Europe and the United States. Thus, an indicator component that would be appropriate in Western studies could lead to distorted estimates in China. Other problems related to transferability are population-specific time-activity patterns, temperature, overall health status, and age distribution in the population. The Chinese epidemiological studies are mainly from Beijing, and there should, in principle, be no serious obstacles to transferring the risk estimates to Shanxi. The SO_2 level in Shanxi cities does not differ substantially from that in Beijing. The regional level of surface ozone is only slightly higher in the Beijing area as compared to Shanxi (own estimates, see Aunan et al., 2000b and reference therein). The climate and seasonal pattern of air pollution is quite similar in the two areas. However, local conditions such as the topography, which gives inversion episodes in winter, could imply higher maximum levels in Shanxi, which may be important for health impacts.

Many epidemiological studies report the associations between air pollution and risk of adverse effects in terms of relative risks or odds ratios, which may be used to derive 'relative functions'. Relative functions indicate the percentage increase in the frequency of a given health effect (often denoted end-point) per $\mu g/m^3$ increase of a given air pollution indicator. In the European ExternE program, it was concluded that quantitative estimates of health effects of air pollution are more reliably transferable between locations if expressed as percentage change (per unit of exposure) rather than as absolute numbers (EC, 1995). This is to ensure that the calculated possible reduction in health damage from a reduction in the population exposure in the applied study is a function of the actual frequency before abatement takes place. This is also the approach taken in the following, where observed or estimated frequencies of the health effect end-points in the area of interest are combined with the relative functions to give 'absolute functions', i.e. the change in number of cases per million inhabitants per $\mu g/m^3$. In other words, this is a way of 'calibrating' the functions.

We relied on Li (2000) for data concerning several of the health indicators necessary to calibrate the exposure–response functions for Shanxi. Data on the frequency of chronic and acute respiratory symptoms and asthma, however, were not available, thus we decided to apply data from a study in Guangzhou in the Guangdong province (Aunan and Li, 1999). In this paper, we report only the health benefit estimated by applying the PM_{10} functions. The functions, shown in Table 4, are linearised and annualised.

To assess the impact of air pollution on mortality, we employ a disaggregated cohort-year approach that allows us to estimate the long-term impacts of reducing mortality rates in different age groups. Studies in the United States have reported a strong association between the long-term level of particles and mortality risk ratios in populations (Pope et al., 1995; Dockery et al, 1993; Abbey et al., 1999). This mortality end-point has, in order to distinguish it from the reported short-time effect, been denoted 'chronic mortality'. The shortening of life-time that a death associated with the long-term level of air pollution causes is likely to be considerably longer than that of a death associated with an acute episode of air pollution (US-EPA, 1996). Chronic exposure thus has a larger impact on life expectancy in the population per $\mu g/m^3$. The abatement options we study will contribute to such an increased life expectancy in the population. Estimates of the annualised impact of this increase can directly be compared with the annualised cost of the options. Because we find it questionable to add 'acute mortality' to 'chronic mortality', we decided not to include the first in our estimates.

In our study, we estimated the effect of changes in the chronic exposure on life expectancy by applying the function proposed in the ExternE project (EC, 1998) (derived from Pope et al., 1995) uniformly on 5-year age groups between the ages of 30 and 90 in the 1998 Chinese life table. A similar method has been recommended earlier in studies by Krewitt et al. (1999) and Brunekreef (1997). In addition, we applied a function for the impact of PM_{10} exposure on infant mortality given in Aunan et al. (2000a) for the youngest age group (0–1 year). This function is based on two studies in the US and in the Czech Republic (Woodruff et al., 1997; Bobak and Leon, 1992). The life table was constructed using 1998 population data (National Bureau of Statistics, 1999b) and standard demographical methodology (Newell, 1988). Data on age-specific mortality rates for China were used due to lack of information for Shanxi. The available data, however, showed that infant mortality rates are lower in Shanxi than China as a whole (National Bureau of Statistics, 1999b). We therefore used the figures for infant mortality in Shanxi in the life table. Compared to using the average figure for China, this increased the estimated life expectancy at birth from 72.2 to 73.5 years.

The estimated increase in life expectancy at birth from a $10\,\mu g/m^3$ reduction in the long-term level of PM_{10} is 0.47 years.[1] The corresponding number of life years gained in different age intervals per person is shown in Fig. 2. These figures were estimated by dividing the estimated numbers of life years gained in each age group by the number of people in the start cohort in the life table. If Δl_i is the number of life years gained per person in age group i, n is the number of age groups, ΔL_i is the increased total number of life years lived in the age group, C is the size of the cohort in the life table, and ΔLE is the increased life expectancy at birth, we have

$$\Delta l_i = \left(\frac{\Delta L_i}{C}\right)$$

[1] Omitting the impact on the infant mortality rate in this calculation, we arrived at 0.42 years per $10\,\mu g/m^3$. As a comparison, a corresponding estimate made for the 1992 life table for Dutch men was about 0.66 years (Brunekreef, 1997).

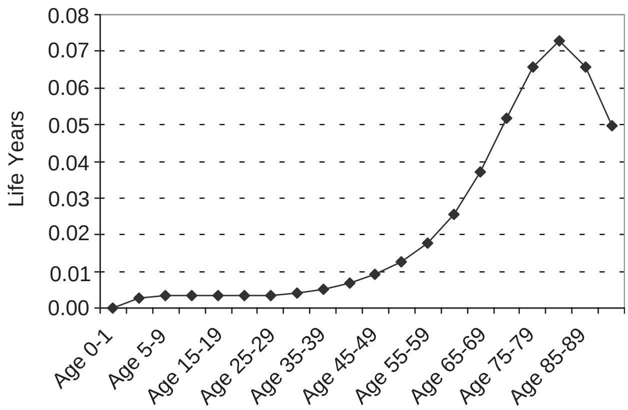

Fig. 2. Age-specific distribution of number of life years gained per person per $10\,\mu g/m^3$ reduction in the long-term level of PM_{10}, as estimated from the 1998 Chinese life table. The total increase in life expectancy at birth (the sum of life years gained per person over all age intervals) amounts to 0.47 years. The integral value of about 0.02 between 1 and 30 years is a result of lower infant mortality only.

and

$$\Delta LE = \sum_{i=1}^{i=n} \left(\frac{\Delta L_i}{C} \right).$$

The largest impact is found in the elderly, and the estimated number of life years gained per person (Δl) per $10\,\mu g/m^3$ for the age intervals 0–50, 50–70, and above 70 was, respectively, 0.05, 0.10 and 0.32 years. For very large PM_{10} reductions, the assumed relationship gives unrealistically large increases in life expectancy. In our case, the maximum increase in life expectancy at birth was estimated to be 1.74 years (as a result of implementing coal washing), which in our view is within a reasonable level.

The number of life years gained was calculated for each cohort (5-year groups) in the present population and for each new cohort consisting of 5 years' of newborns added to the population during the years after abatement measures are assumed to be implemented ('Year 0'). We calculated the number of life years gained in present and future populations for a time period of 90 years, i.e. the time it takes before practically all alive in 'Year 0' are dead. For simplicity, we assumed that the annual number of newborns in urban Shanxi is constant in the future.

The life years gained, ΔLY, due to air pollution abatement in any cohort will be distributed over time, for instance the ΔLY in the age interval 60–65 will occur after 10 years in the cohort that is between 50 and 55 years in 'Year 0'. We consider time after 'Year 0' in intervals ('Year 0–4', 'Year 5–9', etc.) and calculate ΔLY for each time interval, ΔLY_t, where t identifies the interval after 'Year 0'. ΔLY_t thus represents the aggregate number of life years for all cohorts gained in a given period of time (5 years in our case) for all age groups

$$\Delta LY_t = \sum_{i=1}^{i=n} (\Delta LY_{t,i}),$$

where i is the age group, n the number of age groups, and t is the time period (after 'Year 0').

To obtain $\Delta LY_{t,i}$ we calculate $LY_{t,i}$, the number of life years lived in age group i during time period t, with and without the assumption that pollution abatement takes place

$$LY_{t,i} = (C_{t,i} N_i) - LYL_{t,i},$$

where $C_{t,i}$ is the number of people in age group i (at the start of period t), N_i is the number of years in age group i. $LYL_{t,i}$ is the number of life years lost in i in time period t

$$LYL_{t,i} = d_{t,i} N_i a_{t,i},$$

where $d_{t,i}$ is the number of deaths in time period t in age group i and $a_{t,i}$ is the average proportion of time lived in the interval t by those of age group i who die during t. We use $a_{t,i} = 0.5$ except for the first age group, where we use 0.3.

6. Estimates of reduced health impacts

The estimation of how implementation of the abatement options may reduce the average population exposure level in urban areas was based on several assumptions. We assumed that the concentration data that we had were representative for the cities and assumed proportionality between the concentration level and PWE. This is somewhat problematic, even though in this study we have only considered outdoor pollution. The information that was available at this stage concerning monitoring data and population density within the urban areas was, however, too coarse to make more precise estimates.

Moreover, we assumed proportionality between the emissions of the respective component from coal consumption in each source category (see Table 1) and their contribution to the annual PWE. Based on the reasoning given below, we also estimated that 25% of the PWE for PM_{10} and 15% of the PWE for SO_2 would not be influenced by the abatement options. In Shanxi, coal consumption is by far the most important source of SO_2. Moreover, since SO_2 is rapidly converted to sulphate in the atmosphere, the background level of SO_2 in cities is usually not high compared to the concentration caused by local emissions (15% of PWE is slightly lower than the lowest SO_2 level recorded in any of the cities). Long-range transport is generally more prominent for particulate air pollution than for SO_2. For PM_{10} in urban Shanxi, we have used data for source contribution to particulate air pollution in Guangzhou (Aarhus et al., 1999) as a basis for our estimate.

Table 4
Exposure–response functions and economic unit prices applied

Impact (end-point)	Coefficient	Unit price (US$)
Life years lost (all ages)	(see text)	1992[a]
Outpatient visits (OPV)	662 (280–1044)	12.5
Emergency room visit (ERV)	5 (1–8)	12.5
Hospital admissions (HA)	46 (37–58)	2500[b]
Work day loss (WDL) (person-days)	18400 (9200–27600)	1.75
Acute respiratory symptoms in children (ARS-Ch) (person-days)	21500 (14190–32470)	12.5
Acute respiratory symptoms in adults (ARS-Ad) (person-days)	28320 (21130–35520)	12.5
Chronic respiratory symptoms in children (CRS-Ch)	15 (13–18)	3700
Chronic respiratory symptoms in adults (CRS-Ad)	34 (29–39)	3700
Asthma attacks (person-days)	1770 (990–5850)	1

Changes in annual number of cases (or person-days) per million people[c] per 1 $\mu g/m^3$ change in concentration of PM_{10}. Ranges represent tentatively ± 1 SD (see Aunan et al., 2000a).

[a] Not discounted.

[b] About half of the value used by Aunan et al. (2000a) to correct for a shorter average hospital stay in Shanxi than in Guangzhou (Li, 2000).

[c] The share of the population that is <14 years (i.e. children), is incorporated in the functions that applies to adults and children specifically. Thus, the functions can be applied to the total population in an area.

Table 5
Estimated health effects (number of avoided cases) from reduced emission of particles

	Co-generation	Modified boiler design	Boiler replacement	Improved boiler management	Coal washing	Briquetting
Life years lost (1000)	75	2248	2966	879	7723	6088
at ages <0–50>	10	286	376	113	953	758
at ages [50–>	66	1962	2590	767	6770	5330
OPV (million cases)	0.00	0.12	0.15	0.05	0.39	0.31
ERV (1000 cases)	0.03	0.89	1.17	0.35	2.96	2.35
HA (1000 cases)	0.28	8.16	10.72	3.22	27.20	21.64
WDL (million)	0.11	3.26	4.29	1.29	10.88	8.65
ARS-Ch (million cases)	0.13	3.81	5.01	1.50	12.72	10.11
ARS-Ad (million cases)	0.17	5.02	6.60	1.98	16.75	13.32
CRS-Ch (1000 cases)	0.09	2.66	3.50	1.05	8.87	7.06
CRS-Ad (1000 cases)	0.21	6.09	8.01	2.40	20.32	16.16
AA (million cases)	0.01	0.31	0.41	0.12	1.05	0.83

Each measure has been considered separately (see Table 9 in the appendix for the estimated reductions in PWE, and the methodology). Annual figures for all end-points except life years lost, where the figures represent the discounted value of LY lost over a period of 90 years.

The main wind directions during the year (from the south-east during late summer and from the north during winter) and the rainfall pattern during the year (summer monsoon), indicate that the long-range transport of particles into Shanxi is likely to be higher during winter. Since winter is also the season where indigenous emission is the highest, we may assume that long-range transport of particles has a limited impact on the average concentration level in Shanxi cities and towns on an average basis. (Especially in northern areas the influence of long-range transport may, however, be high, due to the vicinity to Beijing). In lack of more detailed information at this stage, the chosen percentages represent in our view acceptable assumptions because we have looked at urban areas in Shanxi on an aggregated level.

By combining the information in Tables 1 and 2, we obtain a rough estimate of the relative potential for reducing outdoor PWE for particles and SO_2 for each abatement option. Details are given in the appendix, Table 9. From these exposure changes and the exposure–response functions in Table 4, the estimated reduction in mortality and morbidity are obtained (Table 5).

7. Valuing health benefits

Although estimates of how air pollution reductions affect PWE and various health indicators are useful, it is advantageous to include an explicit monetary valuation of the health impacts in the analysis. A monetary value of health impacts may easily be compared to monetary costs of investments and costs of CO_2 reduction, providing a common yardstick to assess investments in abatement and energy conservation.

To indicate the main features of the data, we do not distinguish between the importance of effects on children versus adults. As a starting point to obtain a 'value of a life year' (VOLY) lost, we consulted Western studies and surveys of the so-called value of a statistical life (VSL). A VSL is a measure of the economic value of mortality risk to many people, for instance the value to 1000 people of a 1/1000 increase in mortality risk for everyone. Three estimates are given by US-EPA (1997), EC (1998) and NOU (1998). US-EPA (1997) has fitted a Weibull distribution based on estimates from a number of studies in the literature, mostly from the United States. The mean of the distribution, US$4.8 million, is used in their analysis. EC (1998), the large study of externalities of electricity generation, uses US$3.1 million. A recent government commission (NOU, 1998) to set guidelines for cost–benefit evaluations in Norway recommends US$2 million based on a conservative reading of the literature. These estimates amount to ratios of 206, 155 and 70 times GDP/capita, respectively. Miller (2000) provides a survey of such ratios and concludes that a typical value is 120. Although several studies indicate that the ratio of the VSL to GDP per capita is higher in developing countries than in developed countries (e.g. Simon et al., 1999), we have used a ratio from the lower end of the spectrum in this study—i.e. a ratio of 100. As the GDP per capita in Shanxi is about RMB 5040, or about US$630 (National Bureau of Statistics, 1999a), we arrive at a VSL of approximately US$63,000. To transfer this to a unit price for VOLY, we rely on the assessment made by EC (1998), which suggests a VOLY of 98,000. Applying the implied VOLY/VSL-ratio from EC (1998) of 0.0316, we arrive at a Chinese VOLY of US$1992. This value has been applied to all age groups. It corresponds to zero discount rate and a 3% discount rate was applied in the calculation of present value and annualised value of the benefit of reduced mortality rates.

Regarding the unit price of a case of chronic respiratory symptoms, an influential estimate of a case of chronic bronchitis from the United States puts the price at US$260,000 (1990) (US-EPA, 1997). The estimate is actually developed as a fraction of a statistical value of life. Since we downsize the VSL from the US estimates, and since chronic respiratory symptoms include some lighter cases, we start off at US$175,000 in our estimate. We multiply that estimate by the relative GDP/capita ratios of Shanxi and the United States (about 0.021) to account for differences in standards of living and thus in willingness (or ability) to pay.

We assume that acute respiratory symptoms, emergency room visits and outpatient visits can be treated similarly. By adding costs of medicine (approximately 50 RMB), other expenses including round trip taxi and admission fee (approximately 25 RMB) and some willingness-to-pay, we find 100 RMB a reasonable estimate (US$12.50 at an exchange rate of 8 RMB/US$). An alternative procedure to estimating the value of these symptoms could be to adjust Western estimates for the difference in GDP/capita ratios. Western estimates of these symptoms range from US$18 for a case of acute

Table 6
Estimated annualised health benefits (million US$) related to avoidance of the considered health impacts[a]

	Co-gen	Mod. boiler	Boiler replacement	Improved boiler management	Coal washing	Briquetting
Life years lost	4.51	134.33	177.23	52.55	461.55	363.84
at ages <0–50>	0.58	17.08	22.44	6.73	56.96	45.30
at ages [50–>	3.93	117.25	154.79	45.82	404.59	318.54
OPV	0.05	1.47	1.93	0.58	4.89	3.89
ERV	0.00	0.01	0.01	0.00	0.04	0.03
HA	0.69	20.40	26.81	8.04	68.01	54.09
WDL	0.19	5.71	7.51	2.25	19.04	15.15
ARS-Ch	1.62	47.68	62.66	18.80	158.95	126.42
ARS-Ad	2.14	62.81	82.54	24.76	209.37	166.53
CRS-Ch	0.33	9.85	12.94	3.88	32.82	26.11
CRS-Ad	0.77	22.55	29.63	8.89	75.17	59.79
AA	0.01	0.31	0.41	0.12	1.05	0.83
Sum, central	10.32	305.13	401.66	119.88	1030.89	816.68
low_1	7.70	227.45	299.31	89.41	766.65	607.77
$high_1$	13.44	397.79	523.86	156.16	1348.30	1067.09
low_2	4.13	122.05	160.67	47.95	412.36	326.67
$high_2$	25.80	762.82	1004.16	299.70	2577.23	2041.70
low_3	2.79	82.47	108.56	32.40	278.62	220.72
$high_3$	38.18	1128.98	1486.15	443.55	3814.29	3021.72

[a] The three sets of low and high estimates are obtained by using (1) the uncertainty ranges in the dose response functions (Table 4) only; (2) by assuming lognormal distributions with $\sigma_g = 2.5$; and (3) by assuming lognormal distributions with $\sigma_g = 3.7$.

Table 7
Net annual costs of CO_2 reductions

	Abatement cost (US$/ton CO_2)	Local health benefit (US$/ton CO_2)[a]	Net cost of CO_2 reductions (US$/ton CO_2)[a]	Net cost of CO_2 reductions (US$/ton CO_2)[b]
Co-generation	−30.0	32.4	−62.4	−38.7
Modified boiler design	−6.2	23.8	−30.1	−12.7
Boiler replacement	−2.7	32.6	−35.3	−11.5
Improved boiler management	9.2	32.4	−23.2	0.4
Coal washing	22.7	87.3	−64.5	−0.9
Briquetting	27.3	120.4	−93.1	−5.3

[a] Using central estimates of health benefits.
[b] Using lowest estimates of health benefits (low_3).

respiratory symptoms for children (US-EPA, 1997) to US$225 for an emergency room visit (ORNL/RFF, 1992). To adjust these estimates with relative GDP/capita ratios would give very low costs, lower than what medicine and admission fees actually cost in China. However, our estimate is approximately one-third of an estimate from Taiwan (US$40; see Alberini et al., 1997), which we feel is a reasonable adjustment for standards of living.

Our estimate of a hospital admission is based on a study of the average cost of seven respiratory diseases in a hospital of Guangzhou (Aarhus et al., 1999), but adjusted for shorter average stays in hospital in Shanxi (Li, 2000). Our estimate of a workday loss simply equals GDP/day/capita in Shanxi.

The unit price estimates are included in the last column of Table 4. Combining these values with the values in Table 5, the annualised monetised benefits in Table 6 are obtained. The largest share of the benefit relates to reduction in mortality, 44%, whereas acute respiratory symptoms account for 36%. It should be noted that our estimates of economic benefit related to the impact on 'chronic mortality' of abatement options are about half the value (53%) of the estimates obtained when using the 'standard' method, i.e. estimating the annual number of deaths avoided due to reduced chronic exposure (see e.g. EC, 1998) and applying a corresponding VSL (which would be US$63,000 as indicated above). The advantage of our method is that it depicts the age-specific distribution of the impact.

In Fig. 4 in the appendix, the impact on life years lost in the present and future population is shown for one of the abatement measures. This is the discounted impact that forms the basis for calculating the annualised impact. The present population profile in Shanxi is estimated based on data from CITAS (2000). As a result of the present age distribution in Shanxi and the impact of air pollution on remaining life expectancy being considerably higher in the elderly, the lion's share (81%) of the economic benefit of reduced mortality is due to increased number of life years lived at ages >50 years in the population that is alive in the abatement year. About 5% and 8% of the economic benefit are attributable to increased number of life years lived at ages below 50 in the present and the future population, respectively. The remaining 7% of the economic benefit is attributable to increased number of life years lived at ages >50 years in the future cohorts. About 9% of the total economic benefit related to increased life expectancy is due to reduced infant mortality, which increases the number of life years lived also in younger age groups.

In Table 7, we have included the local health benefits per ton CO_2 reduction for the six measures. By subtracting these values from the abatement costs, we obtain net costs of CO_2 reductions (column four). We see that all measures are win–win options in a social sense. Only when applying the lowest estimates for the health benefit, one measure turns out to have a small positive net cost. The results are based on many uncertain assumptions; the uncertainties are discussed in the next section. It should be noted that when net costs are negative, the mutual ranking of different measures according to costs per unit CO_2 reduction becomes less relevant (small reductions imply lower—i.e. more negative—unit cost).

8. Uncertainty considerations

Ideally, an uncertainty estimate should incorporate uncertainties that cascade through all steps of the analysis. It is particularly important to represent uncertainties in parameters to which the final result is specially sensitive. The estimated standard deviations for the exposure–response functions, f_{ER}, are included in Table 4. However, these uncertainties are generally not the most important ones, and typically the benefit estimates are more sensitive to uncertainties in parameters at an early step of the analysis that influences all subsequent steps. A sensitivity test of our results showed that one such critical uncertainty is the assumed PM_{10}/TSP ratio.

Rabl and Spadaro (1999) have given a thorough discussion of the uncertainty terms. They argue that most distributions are close to lognormal, implying the use of multiplicative confidence intervals about the geometric mean μ_g ($\approx$median), e.g. $[\mu_g/\sigma_g,\ \mu_g \cdot \sigma_g]$ for the 68% confidence level. For the uncertainty in estimates of health impacts based on concentration changes in PM_{10}, they suggest that σ_g is about 2.8 with the largest contributions from the dispersion model and from the procedure of transferring exposure–response functions from one region to another ($f_{\text{ER-trans}}$).

The uncertainty ranges given in Table 4 were used to obtain low and high estimates for health effects. By calculating the corresponding monetary values and totalling all low and high estimates, the values denoted low_1 and high_1 in Table 6 are obtained. Since the errors in these coefficients for the different end-points are roughly independent, this gives an overestimate of the contribution of this uncertainty source to the sum of all impacts, but the benefit from reduced mortality is so dominant that the overestimation is small. However, other sources of uncertainty are likely to play a greater role. The values denoted low_2 and high_2 in Table 6 have been obtained by assuming multiplicative confidence intervals (68% confidence level) as recommended by Rabl and Spadaro (1999). We have set σ_g to 2.5, which we believe is reasonable for the estimation of effects on health end-points. This is slightly lower than suggested by Rabl and Spadaro. As above, one may argue that the errors in $f_{\text{ER-trans}}$ for the different impacts are, to some extent, independent. On the other hand, our calculation of concentration changes may seem primitive. However, one should note that the changes depend on the *ratio* of estimated change in PM_{10} emission for the measure to the total PM_{10} emission. If the relative error in one emission factor is the same as in the weighted emission factor for all measures, the concentration change is not affected. Furthermore, to some extent, overestimation in some areas is likely to be compensated by underestimation in others (see Rabl and Spadaro, 1999).

The assumption that 25% of PWE for PM_{10} is not affected by the considered measures is, of course, important, but not critical. Assuming another fraction, say p%, will change all results related to health benefits by $(1-p/100)/0.75$, i.e. with $p = 40\%$ by a factor of 0.8.

Finally, we have included results by assuming σ_g for unit prices to be 2.5, giving a total σ_g of 3.7 (low_3 and high_3 in Table 6). This may be an overestimate at least for some of the unit prices.

In our calculations, we have not considered indoor air pollution separately. Use of unvented coal stoves may cause high levels of indoor air pollution, especially in rural areas. This may contribute to enhanced rates of chronic respiratory disease and lung cancer, as reported from Anhui Province, Yunnan Province and Guangzhou (Pope and Xu, 1993; Mumford et al., 1987; Liu et al., 1993). A study by Wang and Smith (1999a, b) showed that the health benefit was much higher per ton reduction in particulate emissions from household stoves than from coal-fired power plants. In this paper, we have limited the focus to outdoor air pollution and its possible impacts on crude mortality rates, respiratory morbidity and some consequential end-points such as hospital admissions, well aware that indoor pollution may have significant effects on public health in Shanxi Province. However, Li (2000) argues that indoor air pollution is much less serious in urban areas than in rural areas and that lack of sealing causes rapid exchange of indoor and outdoor air in Shanxi.

9. Marginal cost curves

The potential for CO_2 savings available in Shanxi at a negative cost is not quite as large as one would think when viewing each abatement measure in isolation. The reason is that the measures overlap. To account for the overlap, we construct ordinary and social marginal cost curves. The social cost curve includes the value of

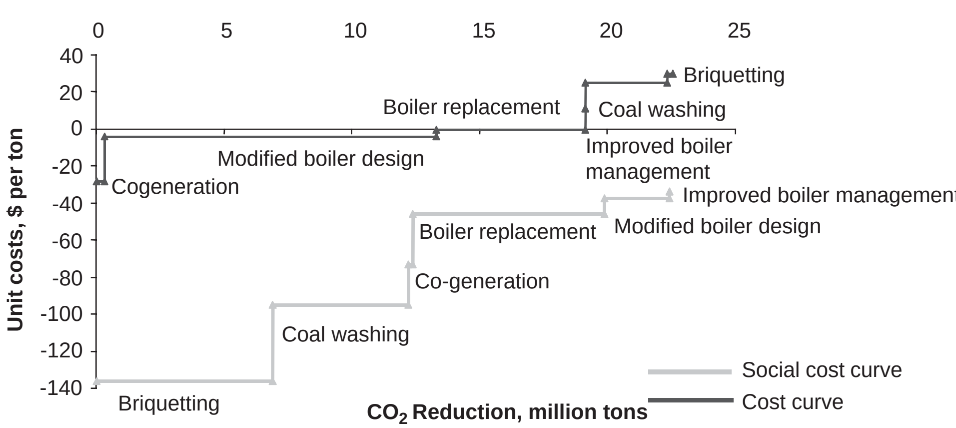

Fig. 3. Marginal cost curve and marginal social cost curve for the six abatement options.

co-benefits, whereas the ordinary cost curve ignores the co-benefits.

To calculate the social and ordinary cost curves, we assume (as has been implicit all along) that all measures of the same kind are identical—they have the same costs and impacts. We also ignore uncertainties. With these assumptions the marginal cost curve and marginal social cost curve in Fig. 3 are obtained. In the marginal cost curve we note, for instance, that the potential for 'boiler replacement' is approximately six million tons of CO_2, which is substantial, but less than the 12 million tons obtained when viewing the measure in isolation (Table 3). The reason is that for some of the boilers it is cheaper to modify the design of old boilers than to replace them. Similarly, but more extremely, improving the management of old boilers, closing leakages and the like, carry a potential of zero since it is better to upgrade and replace boilers. A more detailed analysis of boiler management would probably find pockets of positive potential.

The social marginal cost curve changes the ranking of the measures since the measures have different impacts on local pollution and health. The potential of briquetting increases from 0.2 to 6.8 million tons. The potential of the multilayer combustion system, on the other hand, decreases from 12.8 to 2.5 million tons. We may also note that improving boiler management carries a zero potential in the social sense. These results are clearly very dependent on our assumptions about emission-reduction potentials (Table 2).

The full potential of the six measures considered reaches 22 million tons of CO_2. All of this potential is profitable in the social sense. Nineteen million tons, which corresponds to about half of all CO_2 emissions in Norway, are profitable in a pure economic sense given that credit is available, etc., and thus are win–win options. Whether or not all of this potential could be realized under CDM is another matter.

10. Other environmental benefits

We have included only health effects in our calculations. Reduced concentrations of particles and SO_2 will certainly also have other positive effects. The most important may be reduced corrosion on materials. Several studies in Europe have indicated that lower SO_2 levels considerably reduce replacement and maintenance costs of buildings and other constructions (Kucera et al., 1993; Cowell and ApSimon, 1996). The high levels of particles in the air also lead to soiling of surfaces, as documented, e.g. in a study near Datong (Shanxi) where an annual dust loading up to $1\,\mathrm{kg\,m^{-2}}$ on horizontal surfaces was measured inside a temple grotto (Salmon et al., 1995). The very high SO_2 levels in Shanxi are also likely to harm the vegetation. Since the soils in the region are insensitive to acidification, the effects will be direct rather than through soils. Chameides et al. (1999) have argued that high atmospheric aerosol concentrations may reduce crop yield in China by up to 30%. If this is correct, a considerable loss is certainly to be expected in most of Shanxi. Several studies have indicated that ozone concentrations in parts of China are so high that there may be reductions in yields of some crops, as for instance soybeans, wheat and corn (see e.g. Aunan et al., 2000b). Some reductions may occur in Shanxi (Aunan et al., 2000a). Since the ozone concentration increases with increasing NO_x emissions, a large increase in NO_x emissions should be avoided. However, measures only in Shanxi will have limited effects; concerted actions in larger regions are necessary.

11. Discussion and conclusions

In spite of the preliminary nature of this study, some conclusions may be drawn. As seen from Table 7, three of the selected measures are win–win options, with negative abatement costs even without considering health and other environmental benefits. Only when using the lowest estimates of health benefits (low_3) included in Table 6, which are probably too conservative, do the net costs become slightly positive for one measure. Our results also show that rankings of abatement measures according to cost effectiveness are substantially altered when local health and environmental benefits are taken into account. For example, briquetting is one of the most expensive options for reducing emissions of CO_2, but is one of the most cost-effective options for providing local benefits. In spite of the large uncertainties, our calculations thus clearly underscore the importance of including co-benefits in analyses of costs of CO_2 mitigation. Conversely, if the focus of analysis is on local problems, then reduced GHG emissions can be seen as a 'bonus'.

In this paper, we have not explored the potential practical, physical and institutional barriers to implementation of the measures. For instance, water shortage and water pollution are major problems in Shanxi, which make coal washing less attractive. This could imply that we have overstated the potential for this measure. The use of briquettes tends to make boiler management more complicated. Conversely, we may have underestimated the potential of co-generation by assuming that only paper and textile industries are viable venues for this measure.

Our simplified approach to estimating population exposures did not take full account of the differences between and within the various cities. It would be desirable to identify sources of air pollution with greater detail and precision to improve the analysis of the impact that abatement is likely to have on population exposure. Indoor air pollution must also be taken into consideration, as it is a main source of exposure to many people. In the case of increased use of high-quality briquettes in rural households, the benefit of this measure could be higher than our estimates show because of effects on indoor air quality (see e.g. Wang and Smith, 1999a, b).

A continued and intensified emphasis on measures that increase energy efficiency and on the use of clean coal technologies seems, in our view, essential. Concern has been voiced that measures that improve the performance of coal in energy generation constitute a cul-de-sac, contributing to a lock-in of coal technologies (McDonald, 1999; Unruh, 2000). However, considering the state of energy resources in China and the lack of progress in the rest of the world with respect to the development of economically viable, renewable energy technologies, it does not seem realistic for China to break away from coal for the time being.

The policy implications of recognising that many GHG mitigation measures provide substantial local health and environmental benefits are profound. An integrated approach to analysing the costs and benefits of alternative mitigation measures would identify the options with the largest synergies, and may be a key to increasing the interest of developing countries in GHG mitigation. There is little doubt that the future of the Kyoto Protocol depends to a large extent on involvement by developing countries. So does the future growth rate of the world's CO_2 emissions.

Finally, this study has implications for discussions of rules regulating CDM projects. We take note of the controversy over 'additionality' requirements for eligibility, which means that for a project to be registered and credited under CDM, it must not be viable in the absence of CDM. This is a double-edged sword. It may certainly prevent provision of credits for fake emission reductions. On the other hand, this requirement may limit possibilities for funding some of the most locally and regionally advantageous projects under CDM. In our view, it is important that measures related to clean coal technologies and energy efficiency, such as those discussed in this paper, are considered eligible under CDM in part because of the substantial local co-benefits. The prospect for such projects seems good judging from the Bonn (July 2001) agreement, which established a fast-track procedure for small-scale CDM projects, whereby energy efficiency improvement projects, among other types, can benefit from simplified modalities and procedures (see e.g. Torvanger, 2001). Large co-benefits in terms of reduced local and regional air pollution could, moreover, in our view easily be seen as an indicator of whether a CDM project fulfils the dual objective of this mechanism, namely, that it shall assist the host country in achieving sustainable development in addition to mitigating GHG emissions.

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Appendix A

For details see Tables 8 and 9 and Fig. 4.

Table 8
Emission factors

	EF (TSP) (kg/ton)	EF (SO_2) (kg/ton)
Households		
Rural	20	20.2
Urban	20	20.2
Industry		
Power	3.6	18.1
Coke making	2.9	5.0
Industrial combustion	8.0	18.1

Table 9
Potential reduction in population weighted exposure (ΔPWE)[a]

	$\Delta PWEPM_{10}$[b] ($\mu g/m^3$)	$\Delta PWESO_2$[c] ($\mu g/m^3$)
Co-generation	0.4	0.3
Modified boiler design	10.9	11.1
Boiler replacement	14.4	10.7
Improved boiler management	4.3	3.2
Coal washing	36.4	40.9
Briquetting	29.0	20.6

[a] By combining the information in Tables 1 and 2, we obtain a rough estimate of the relative potential of reducing outdoor PWE for particles and SO_2 for each abatement option:

$$\Delta PWE_{i,j} = \Sigma_k (S_k E_{j,k} R_j)\, PWE_i,$$

where $\Delta PWE_{i,j}$ is the reduced population weighted exposure (PWE) for component i for abatement option j, PWE_i the PWE for component i before abatement (adjusted for the share that is assumed inert to emission reductions in the coal sector in Shanxi, see below). S_k the relative contribution of source k feasible for regulation by option j to PWE (based on Table 1), $E_{j,k}$ the share of source k feasible for regulation by option j (see Table 2), R_j the reduction potential of abatement option j (see Table 2).

[b] Assuming that 25% of PM_{10} is not caused by use of coal in Shanxi, i.e. PWE is $210\,\mu g/m^3$.

[c] Assuming that 15% of SO_2 is not caused by use of coal in Shanxi, i.e. PWE is $158\,\mu g/m^3$.

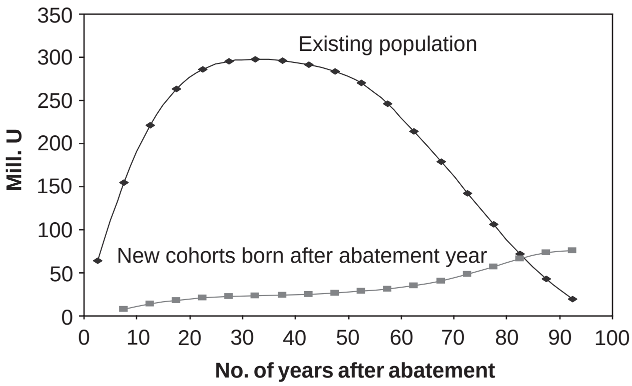

Fig. 4. Present value of the life years gained in urban Shanxi estimated for the abatement option 'Modified boiler design' ($\Delta PWE = 10.7\,\mu g/m^3$).

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