Co-benefits from CO_2-emission reduction measures in Shanxi, China – their importance in context of CDM

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'Policy Reform and the Environment in China'
Conference organised by PACE (Professional Association for China's Environment)
August 31 – September 2, 2000, Washington DC:

Abstract

This paper analyses a set of CO_2-reducing abatement options related to coal consumption in Shanxi, China. The costs and potential for abatement are investigated for different economic sectors, and the entailed emission reductions are estimated in terms of CO_2, SO_2, and particles. The present 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 population exposure resulting from the abatement measures. By means of exposure-response functions from Chinese and international epidemiological studies, the health effects from implementing the measures are indicated. An economic evaluation of the reduced health effect is made by employing unit prices of health impacts based on the damage cost approach. This assessment of CO_2-reducing abatement options in Shanxi demonstrates that the measures are profitable in a socioeconomic sense.

Introduction

The flexibility mechanisms in the Kyoto Protocol give the countries with emission targets an incentive to meet their obligations in a cost-effective manner. One of these mechanisms, defined in Article 12 in the Protocol, is the *Clean Development Mechanism* (CDM). Through CDM, developed countries (Annex I countries to the Framework Convention) may invest in greenhouse gas (GHG) mitigation projects in developing countries (non-Annex I countries) in order to be credited for the reductions. This gives the developing countries an economic incentive to reduce GHG emissions, and encourages them to take an active part in the global process of reducing the growth of greenhouse gas emission.

For several reasons, which we will not go into in this paper, many developing countries are sceptical to the idea that Annex 1 countries should be allowed to fulfil part of their obligation in developing countries. The purpose of this paper, which describes the results of a case study in the main coal producing province in China, Shanxi, is to draw attention to the possibly large co-benefits that implementation of CDM projects may have in the host country. These co-benefits relate to the improved air quality that may result from GHG mitigation measures because important sources of GHG, such as power production and the transport sector, also are main sources of particulates, SO_2, NO_x, and various hydrocarbons. Our study indicates that CDM projects may be beneficial not only to the involved enterprises, but also to the local society where these enterprises are situated. Although detailed rules on verification, reporting and accountability for CDM and the other flexibility mechanisms in the Kyoto Protocol are still not agreed upon, it seems clear that technological transfer will be obligatory under the CDM. Updating of old and polluting technology is essential for a sustainable development in many parts of the developing world, and is likely to improve people's quality of life.

Whereas the climate change issue may not be given high priority on the political agenda, in many developing countries there is now an increasing focus on local and regional pollution problems. The Chinese government realises the significance of urban pollution, and in November 1999 announced the 'Clean Energy Action', which aims at having most big cities satisfy the national class 2 standards within 3 – 5 years. Regulating the use of leaded gasoline in major cities and requiring all car manufacturers in China to install catalytic converters and electric ignition systems in new cars clearly indicates a policy shift (Gan, 1999).

Synergies between abatement options addressing different environmental problems

The main co-benefits (also called secondary or ancillary benefits) of GHG abatement related to improved air quality 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. An important characteristic of the co-benefits, as opposed to the benefits related directly to reducing the GHGs, is that they appear within a relatively short time after the emissions are reduced. The importance of including co-benefits in socio-

economic analyses of GHG abatement was recently demonstrated in a case study in Hungary (Aunan et al., 1998; Aaheim et al., 1999).

Generally, the largest local and regional air pollution problems are found in countries that do not have emission reduction obligations in the Kyoto Protocol. For instance there are large air pollution problems in many megacities in developing countries, often due to a heavy reliance on coal in power production and outdated technology in automobiles. In China 72% of the energy is generated from coal, 18.5% from oil and 2.4% from natural gas (1998 figures). The share of coal has slightly decreased the last decades, the share of oil and gas has fluctuated, whereas the share of hydro-power has been increasing and now amounts to 7.1% of the production (National Bureau of Statistics, 1999). The concentrations of SO_2 and particulates in air violate the WHO air quality guidelines severalfold in many cities. Because typical '1. generation' air pollution abatement measures, as for instance 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. Several studies have demonstrated that CO_2 reductions in these countries may be accompanied by substantial reductions in health damage, e.g. in premature mortality and respiratory diseases (Lee Davis et al., 1997, Wang and Smith, 1999a and b, Dessus and O'Connor, 1999; Ekins, 1996).

There are, however, not necessarily large synergies between local and global environmental targets (Eskeland and Xie, 1998). An abatement strategy that is cost-effective with respect to a global perspective may therefore not be cost-effective in a local or regional perspective. Generally speaking, the measures that have the largest potentials for synergies are those that lead to a shift towards less carbon-intensive fuels, and measures that simply reduce the use of fossil fuel *per se*, as energy efficiency measures (assuming the demand is not increased) and introduction of renewables. Abatement measures devised specifically towards reducing SO_2 and particulates, i.e. typical '1. generation measures', usually do not have a significant impact on GHG emissions.

In this study we have sought to identify viable options for reducing emissions of GHG in the Shanxi province in China. By estimating the local and regional air pollution effects of implementing abatement measures, 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, building materials and cultural heritage are also included. In the first phase, documented in this article, we have focused on analysing the environmental impacts of six different abatement options that mainly are applicable in the industry sector, the power sector and in rural households. The results are preliminary, and 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).

Shanxi province

Shanxi province lies in the central parts of North China covering an area of 156,300 square km 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 mean annual precipitation is between approximately 350 and 700 mm, the amount increasing gradually from northwest to the southeast. January, the coldest month, has average temperatures of -1°C to -15°C, whereas the average temperatures of July, the hottest month, are between 20°C and 27°C.

The province contains 118 counties with its capital set in Taiyuan City. The total population of Shanxi is 31.7 mill. (1998), of this we estimate that about 16.2 mill. 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 percent 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 the most important industry and the coal production in Shanxi represents about 26% of the total production in China (354 Mtons in 1997). 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 (235 Mtons in 1997). Cokemaking plants process more than 50% of the coal retained and the coke production in 1997 was about 53 Mtons (weight as coke). The coke production in Shanxi represents nearly half of the Chinese production. About two thirds of the coke produced in Shanxi is normally exported to other provinces and countries. In 1997 coke from Shanxi accounted for about 55% of the total export from China (China stands for about 60% of the world trade of coke). The consumption of coke within the province is to a large extent in metallurgical industry.

Heavy industry in Shanxi includes production of machinery, e.g. tractors, locomotives, automobiles and equipment used in mines and metallurgical industry. Main light industry sectors are textile, paper and food. As compared to some of the coastal provinces, which after the reform starting in 1987 have benefited from a favorable economic policy and have had a substantial economic growth, Shanxi has, together with other central and western provinces, been lagging behind. In 1998 the GDP/cap in Shanxi was 79% of the average in China (National Bureau of Statistics, 1999).

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 to this comes the widespread use of coal in small coal-fired commercial boilers for heating and steam generation and 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. Based on these figures, we have estimated the total emission of TSP (total suspended particulates) and SO_2 from coal consumption in Shanxi (Table 1, two last columns). Emission factors

for TSP range from 2.9 kg/ton coal for the coke making industry to 20 kg/ton coal for households. For SO_2 the values range from 5.0 kg/ton coal (coke making) to 20.2 kg/ton coal (households). The low SO_2 emission factor for coke making reflects both the generally lower sulfur content in coal used for coke making and that much of the coke is exported from the province (some sulfur is retained in the coke). Details are given in Table A1 in the Appendix. Our estimate of total SO_2 emission is somewhat higher than given by National Bureau of Statistics (1999).

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

	Consumption (Mill. tons raw coal)	% of total	TSP (1000 tons)	SO_2 (1000 tons)
Total	**136.15**	**100.0**	**667.8**	**1532.7**[1]
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

[1]*This figure is somewhat higher than other estimates, eg. by National Bureau of Statistics (1999) and Streets and Waldhoff (2000).*

The air pollution situation is severe in many cities of Shanxi, 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 (see Figure 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), Figure 1 shows that this threshold is exceeded severalfold 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 indicate. 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 periods of inversion and stagnant air masses during winter time, thus enhancing the concentration of air pollutants. The main emission sources in Taiyuan are two co-generation power plants, with installed capacity of 1000 and 600 MW, respectively, and Taiyuan Iron and Steel Company, which is one of the major iron and steel complexes in China. In Taiyuan these three sources represent about 67% of the total non-residential emissions and are all

situated in central areas of the city. Taiyuan also has a substantial cement production. The main emission sources from iron and steel production relate to coke production (for own use), power production for own electicity 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. Datong city also has machine-building and building-materials industries, and a substantial cement production.

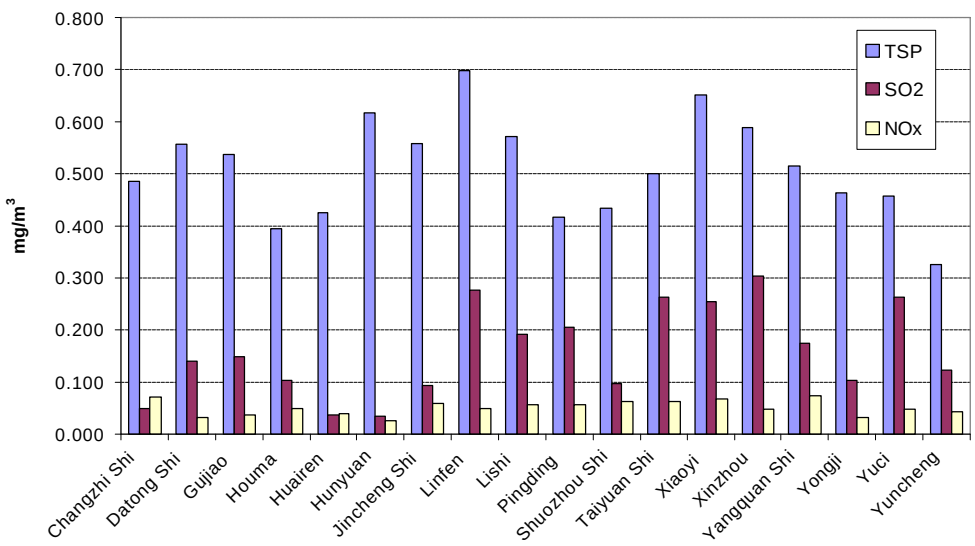

Figure 1. Air quality in some cities in Shanxi (annual average for 1997 and 1998). Source: Shanxi EPB, 1999. For air quality guidelines: See text.

A population weighted exposure (PWE) can be calculated from:

$$\text{PWE} = \frac{1}{P}\sum_i c_i \cdot p_i$$

where p_i is the population in city i, c_i is the average level of the pollutant in city i, and P is the total population in all cities. Again assuming PM_{10} = 0.55 TSP, PWE for PM_{10} in the 18 cities is 280 μg/m^3; the corresponding value for SO_2 is 186 μg/m^3. In the calculation of weighted exposure we assumed that the population in the cities for which we did not have population data, was 75.000 (the population database (UN, 1998) contained population in cities with 100,000 and more inhabitants).

Abatement measures

Six different abatement options that will reduce the emissions related to use of coal were analysed according to costs and reduction potentials (CO_2, TSP and SO_2). The options

and the amount and share of the present coal consumption that are assumed eligible for each option are listed in Table 2. We have looked at options feasible in the industry sector, the power sector and in rural households.

The total coal consumption is about 136 mill. ton (average for 1995-97, see Table 1). The share of the source consumption that is eligible to the abatement options is assumed to be between 60 and 100% for most options, except co-generation (Table 2). This option is assumed feasible at the given conditions only in the paper and textile industry, which has 2.3% of the industrial coal consumption. The assumed emission reduction potentials, as share of the baseline emission coefficient, are given in Table 2, while abatement cost per CO_2 unit and the reduction potentials for various components are rendered in Table 3. In the following calculations we have considered each measure separately. Since some of the abatement measures relate to the same sources, the emission reductions will in practice depend on whether other measures already have been implemented.

Table 2. Potentials for the abatement options. Amount (mill. tons of raw coal consumed), percentage share of source consumption eligible to the selected abatement options, and emission reduction potential (as percentage of baseline emission coefficient).

	Rural house-holds	Power	Industrial combustion	Emission reduction potential (%)		
				CO_2	TSP	SO_2
Coal washing (mill. tons)		23.8	29.9	10	35	40
% of source		84.1	100.0			
Briquetting (mill. tons)	1.0		29.9	10	35	35[3]
% of source	100.0		100.0			
Improved management (mill. tons)			22.4	8	8	8
% of source			75.0			
Boiler replacement (mill. tons)			22.4	25	25	25
% of source			75.0			
Co-generation (mill. tons)[1]			0.7	21	21	21
% of source			2.3			
Modified boiler design (mill. tons)[2]		17.0	17.9	17	17	17
% of source		60.0	60.0			

[1] *Paper and textile industries only.*
[2] *Multilayer combustion system.*
[3] *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%.*

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

	Abatement cost (US$/ton CO_2)	CO_2 (mill. 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

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 USA when necessary (Aunan et al., 2000a). 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 mill. inhabitants per $\mu g/m^3$. In other words, this is a way of 'calibrating' the functions.

Concerning several of the health indicators necessary to calibrate the exposure-response functions for Shanxi we relied on Li (2000). Data on the frequency of chronic and acute respiratory symptoms and asthma were, however, not available, and 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. In an economic evaluation it has to be taken into consideration that some health effects may last for a certain period of time. The basis for the functions is given in Aunan et al. (2000a).

Compared to studies in Europe and the USA, 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. One important exception is the function for

acute mortality and SO_2, where the Chinese studies tend to report a steeper relationship, especially at lower concentration ranges (a log-normal function is suggested in some studies).

There are several problems connected to transferring risk estimates from one population to another. For instance the composition of the car fleet in Western Europe and USA, where most of the epidemiological studies have been performed, differs substantially from that in China. This, together with other differences as the widespread use of coal in China, implies that the air pollution mixture (co-pollutants) differs substantially between China and Western Europe and USA. Thus, an indicator component found to be suitable in western studies, could lead to biased 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 for transferring the risk estimates to Shanxi. The climate and seasonal pattern of air pollution are quite similar in the two areas, however, local conditions as the topography giving inversion episodes in winter could imply higher maximum levels in Shanxi, which may be important for health impacts. Concerning the air pollution mixture in Beijing and Shanxi (co-pollutants) the SO_2 level in Shanxi cities does not differ substantially from that in Beijing. The regional ozone level is only slightly higher in the Beijing area as compared to Shanxi (own estimates, see Aunan et al., 2000b and reference therein).

Table 4 Exposure-response functions. Changes in annual number of cases (or person-days) per million people (all ages) per 1 $\mu g/m^3$ change in concentration of PM_{10}. Ranges represent tentatively –1 SD (see Aunan et al., 2000a).

Impact (End-point)	Coefficient	Unit price ($)
Premature deaths (adults)	2.2 (0-4.1)	63,000
Infant deaths	1.2 (0.8-1.7)	63,000
Outpatient visits (OPV)	662 (280-1,044)	12.5
Emergency room visit (ERV)	5 (1-8)	12.5
Hospital admissions (HA)	46 (37-58)	2,500[1]
Work day loss (WDL) (person-days)	18,400 (9,200-27,600)	1.75
Acute respiratory symptoms in children (ARS-Ch) (person-days)	21,500 (14,190-32,470)	12.5
Acute respiratory symptoms in adults (ARS-Ad) (person-days)	28,320 (21,130-35,520)	12.5
Chronic respiratory symptoms in children (CRS-Ch)	15 (13-18)	4,700
Chronic respiratory symptoms in adults (CRS-Ad)	34 (29-39)	4,700
Asthma attacks (person-days)	1,770 (990-5,850)	1

[1]*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).*

Estimates of reduced health impacts by implementing abatement measures

Several assumptions must be made to estimate how implementation of the abatement options may reduce the average population exposure level in urban areas. We assumed that the concentration data that we had were representative for the cities and also proportionality between the concentration level and population-weighted exposure (PWE). This is somewhat problematic, even though we in this study have limited this concept to outdoor pollution. The information that was available at this stage concerning monitoring stations and population density within the urban areas was, however, too coarse to make more precise estimates.

Further, we assumed proportionality between the emission 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 sulfate 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 that the lowest SO_2 level recorded in any of the cities.) For PM_{10} we have used data for source contribution to particulate air pollution in Guangzhou (Aarhus et al., 1999) as a basis for our estimate. The main wind directions during the year (from south-east during late summer and from 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 time is also the season where the indigenous emission is the highest, we 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 parts 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. To some extent overestimating in some areas is likely to be compensated by underestimating in others (see Rabl and Spadaro, 1999).

By combining the information in Table 1 and Table 2 we may 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 A2. From these concentration changes and the exposure-response functions in Table 4, the estimated reduction in mortality and morbidity may be obtained (Table 5). An extended table including lower and upper estimates is given in the appendix (Table A3).

Table 5. Estimated health effects (annual number of avoided cases) from reduced emission of particles. Each measure has been considered separately[1].

	Co-generation	Modified boiler design	Boiler replacement	Improved boiler management	Coal washing	Briquetting
Deaths (adults)	13	376	494	148	1,253	996
Infant deaths	7	208	273	82	692	550
OPV (mill. cases)	0.00	0.12	0.15	0.05	0.38	0.31
ERV (1000 cases)	0.03	0.87	1.14	0.34	2.90	2.31
HA (1000 cases)	0.27	8.00	10.51	3.15	26.67	21.22
WDL (mill.)	0.11	3.20	4.21	1.26	10.67	8.49
ARS-Ch (mill. cases)	0.13	3.74	4.92	1.47	12.47	9.92
ARS-Ad (mill. cases)	0.17	4.93	6.47	1.94	16.42	13.06
CRS-Ch (1000 cases)	0.09	2.61	3.43	1.03	8.70	6.92
CRS-Ad (1000 cases)	0.20	5.98	7.85	2.36	19.92	15.84
AA (mill. cases)	0.01	0.31	0.40	0.12	1.03	0.82

[1]*See Table A2 in the Appendix for the estimated reductions in PWE, and the methodology.*

Valuing health benefits

Although estimates of how air pollution reductions affect PWE and different 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.

In order to indicate the main features of the data we do not distinguish between the importance of effects on children versus adults. We also assume that acute respiratory symptoms, emergency room visits and outpatient visits can be treated similarly. By adding medicine cost (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. Western estimates of these symptoms go from $18 for a case of acute respiratory symptoms for children (US-EPA, 1997) to $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 medicine and admission fees actually cost in China. However, our estimate is approximately one third of an estimate from Taiwan ($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 work day loss simply equals GDP/capita in Shanxi.

To arrive at our preferred price estimate of premature deaths we consulted Western studies and surveys of the so-called value of a statistical life (VSL). (VSL is actually based on a value of a small change in risk.) 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 USA. The mean of the distribution, $ 4.8 million, is used in their analysis. EC (1998) is a large study of externalities of electricity generation that uses $3.1 million. A recent Government Commission (NOU, 1998) setting down guidelines for cost-benefit evaluations in Norway recommends $2 million based on a conservative reading of the literature. Recalculated to VSL/GDP/capita these estimates amount to 206, 155 and 70 respectively. Although several studies indicate that the ratio of the value of a statistical life 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, that is a ratio of 100.

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

The unit price estimates are included in the last column of Table 4. Combining these values with the values in Table 5 the monetized benefits in Table 6 are obtained. The largest benefits relate to reductions in mortality, hospital admissions and respiratory symptoms.

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. The results are based on many uncertain assumptions; the uncertainties are discussed in the next section.

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 that the final result are especially sensitive to. Concerning the exposure-response functions, f_{ER}, the estimated standard deviations are included in Table 4, and these were used to derive the estimates for the health benefit denoted low_1 and $high_1$ as given in Table 6. However, in general these uncertainties are 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 Spandaro (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 on region to another ($f_{ER\text{-}trans}$).

Table A3 in the appendix includes low and high estimates for health effects based only on the ranges in Table 4, i.e. the uncertainty in f_{ER}. By calculating the corresponding monetary values and summing all low estimates and all 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. However, other uncertainty sources are likely to be more important. 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 Spandaro (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 Spandaro. However, as above one may argue that the errors in $f_{ER\text{-}trans}$ 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 TSP emission for the measure to the total TSP 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. The assumption that 25% of PWE for PM_{10} is not affected by the considered measures is of cause 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 i.a. Anhui province, Yunnan province and Guangzhou (Pope and Xu, 1993, Mumford et al., 1987; Liu, 1993). A study by Wang and Smith (1999a, b) showed that the health benefit was much higher per ton reduction in particulates emissions from household stoves than from coal-fired power plants. In this report we have limited the focus to outdoor air pollution and its possible impacts on crude mortality rates, respiratory morbidity and some consequential end-points like hospital admissions, well aware that indoor pollution may have significant effects on public health in this 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.

Table 6. Estimated health benefits (mill. US$) for the considered health impacts[1].

	Co-gen	Mod. boiler	Boiler replacem.	Improved boiler management	Coal washing	Briquet-ting
Deaths (adults)	0.80	23.67	31.11	9.33	78.91	62.76
Infant deaths	0.44	13.07	17.18	5.15	43.58	34.66
OPV	0.05	1.44	1.89	0.57	4.80	3.82
ERV	0.00	0.01	0.01	0.00	0.04	0.03
HA	0.68	20.01	26.29	7.89	66.69	53.04
WDL	0.19	5.60	7.36	2.21	18.67	14.85
ARS-Ch	1.59	46.75	61.44	18.43	155.85	123.96
ARS-Ad	2.09	61.59	80.93	24.28	205.30	163.29
CRS-Ch	0.42	12.26	16.12	4.83	40.88	32.52
CRS-Ad	0.95	28.09	36.91	11.07	93.62	74.46
AA	0.01	0.31	0.40	0.12	1.03	0.82
SUM, central	**8.78**	**239.19**	**309.42**	**98.75**	**335.75**	**803.17**
low_1	4.64	136.59	179.49	53.85	455.33	362.16
$high_1$	10.02	294.64	387.17	116.15	982.16	781.19
low_2	2.9	85.1	111.9	33.6	283.7	225.7
$high_2$	18.1	532.0	699.1	209.7	1773.4	1410.5
low_3	2.0	57.5	75.6	22.7	191.7	152.5
$high_3$	26.8	787.4	1034.6	310.4	2624.7	2087.6

[1]*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* $s_g = 2.5$*; and 3) by assuming lognormal distributions with* $s_g = 3.7$*.*

Table 7. Net costs of CO_2 reductions.

	Abatement cost (US$/ton CO_2)	Local health benefit (US$/ton$CO_2$)[1]	Net cost of CO_2 reductions (US$/ton$CO_2$)[1]	Net cost of CO_2 reductions (US$/ton$CO_2$)[2]
Co-generation	-30.0	22.7	-52.7	-36.1
Modified boiler design	-6.3	16.6	-22.9	-10.7
Boiler replacement	-2.6	22.7	-25.4	-8.9
Improved boiler management	9.2	22.7	-13.5	3.1
Coal washing	22.7	60.1	-37.3	6.5
Briquetting	27.3	83.2	-55.9	4.8

[1]*Using central estimates of health benefits..*
[2]*Using lowest estimates of health benefits (low_3 and $high_3$).*

Other environmental benefits

We have only included 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 considerable reductions in replacement and maintenance costs of buildings and other constructions with reduced SO_2 levels (Kucera et al., 1993 and Cowell and ApSimon, 1996). The

extremely high SO_2 levels are also likely to have harmful effects on 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). Reductions of some percent may occur in Shanxi (Aunan et al. 2000a). Since the ozone concentration increases with increasing NOx emissions, a large increase in NOx emissions should be avoided. However, measures only in Shanxi will have limited effects; concerted actions in larger regions are necessary.

The high levels of particles in the air also lead to soiling of surfaces. This is illustrated in a study in a small village of Yungang, 16 km west of Datong, situated close to a coal-haul highway. Here it was measured an annual dust loading up to 1 kg m^{-2} on horizontal surfaces *inside* a Buddhist cave temple grotto (Salmon et al., 1995). Annual average outdoor TSP in the area was 508 $\mu g/m^3$, and the indoor TSP level in the grottoes was about 300 $\mu g/m^3$.

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 have negative abatement costs and thus are profitable even without considering reduction in health and other environmental damage (win-win options). Using the central estimate of health benefits, the net cost becomes negative for all measures, i.e. all options are profitable in a social sense. Using the lowest estimates (low_3) included in Table 6, which are probably too conservative, the net costs of improved boiler management, coal washing, and briquetting become positive, although small. Our results also show that the ranking of abatement measures according to their cost-effectiveness becomes substantially altered when taking into account the co-benefit, in this case the health benefit, of the measures. One outcome is that briquetting, which from a pure CO_2 perspective seems the most expensive, turns out to be one of the most profitable. In spite of the large uncertainties our calculations thus clearly illustrate the great importance of including co-benefits in estimates of costs of CO_2-mitigation. If the focus is on local problems, the reduced greenhouse emissions can accordingly be seen as a ‘bonus’.

In this paper we have not explored the potential physical and institutional barriers to implementation of the measures that may exist. 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. Concerning co-generation, we may, on the other hand, have underestimated the potential by assuming that only the paper and textile industry is eligible to this measure.

A simplified approach was taken to estimate population exposure, which levels out the large differences between the various cities. There is certainly a need to identify the source contribution to air pollution levels in a more detailed geographical scale, and subsequently to which extent the different abatement options will have an impact on the population exposure. Indoor air pollution must 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 it would directly influence the indoor air quality (see e.g. Wang and Smith, 1999a and b).

The policy implications of realizing that many GHG mitigation measures have large co-benefits, are profound. An integrated approach in analyses of the costs and benefits of alternative mitigation measures will identify the options with the largest synergies, and may be a key to achieving increased interest in greenhouse gas mitigation in developing countries. 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.

Rules for regulating CDM projects is one of the main issues to be agreed upon at the sixth Conference of the Parties (COP6) which is to be held in November this year. Current suggestions set forth an additionality requirement for eligibility. This means that for a project to be registered under CDM, it must be unlikely that it would be implemented in the absence of CDM. This is a double-edged sword. It may certainly prevent crediting for fake emission reductions, but on the other hand it may also limit the possibilities for getting some of the most advantageous projects registered under CDM. In our view, it would do more harm than good if measures related to clean coal technologies and energy efficiency, such as those discussed in this paper, were to be ruled out.

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Appendix

Table A1. 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
Industr. combustion	8.0	18.1

Sources: Handbook of China Environmental Protection (1997), and Fang Jinghua (pers. comm.).

Table A2. Potential reduction in population weighted exposure (DPWE).[1]

	$\Delta PWE_{PM_{10}}$-I[2] ($\mu g/m^3$)	$\Delta PWE_{PM_{10}}$-II[3] ($\mu g/m^3$)	ΔPWE_{SO_2}[4] ($\mu g/m^3$)
Co-generation	0.4	0.3	0.3
Mod. boiler design	10.7	8.6	11.1
Boiler replacement	14.1	11.3	10.5
Imp. boiler management	4.2	3.4	3.3
Coal washing	35.7	28.6	40.1
Briquetting	28.4	22.7	20.2

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

$$DPWE_{i,j} = \sum_k (S_k \cdot E_{j,k} \cdot R_j) \cdot PWE_i$$

where:

$DPWE_{i,j}$ = reduced population weighted exposure (PWE) for component *i* for abatement option *j*

$PWE_{i,}$ = 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 = relative contribution of source *k* eligible to option *j* to PWE (based on Table 1)

$E_{j,k}$ = share of source *k* eligible to option *j* (see Table 2)

R_j = reduction potential of abatement option *j* (see Table 2)

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

[3] Assuming that 40% of PM_{10} is not caused by use of coal in Shanxi, i.e. PWE is 168 $\mu g/m^3$.

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

Table A3 Estimated health effects (annual number of avoided cases) from reduced emission of particles obtainable from implementing the abatement options separately. Low and high estimates only relate to uncertainties in the dose-response functions.

	Co-generation	Modified boiler design	Boiler replacement	Improved boiler management	Coal washing	Briquetting
Mortality (M)	13	376	494	148	1,253	996
Low	0	0	0	0	0	0
High	24	713	937	281	2,377	1,891
Infant mortality	7	208	273	82	692	550
Low	4	130	171	51	434	345
High	10	285	374	112	950	755
OPV (mill. cases)	0.00	0.12	0.15	0.05	0.38	0.31
Low	0.00	0.05	0.06	0.02	0.16	0.13
High	0.01	0.18	0.24	0.07	0.61	0.48
ERV (1000 cases)	0.03	0.87	1.14	0.34	2.90	2.31
Low	0.01	0.17	0.23	0.07	0.58	0.46
High	0.05	1.39	1.83	0.55	4.64	3.69
HA (1000 cases)	0.27	8.00	10.51	3.15	26.67	21.22
Low	0.18	5.39	7.09	2.13	17.98	14.30
High	0.34	10.09	13.26	3.98	33.63	26.75
WDL (mill.)	0.11	3.20	4.21	1.26	10.67	8.49
Low	0.05	1.60	2.10	0.63	5.33	4.24
High	0.16	4.80	6.31	1.89	16.00	12.73
ARS-Ch (mill. cases)	0.13	3.74	4.92	1.47	12.47	9.92
Low	0.08	2.47	3.24	0.97	8.23	6.54
High	0.19	5.65	7.42	2.23	18.83	14.98
ARS-Ad (mill. cases)	0.17	4.93	6.47	1.94	16.42	13.06
Low	0.12	3.68	4.83	1.45	12.25	9.75
High	0.21	6.18	8.12	2.44	20.60	16.38
CRS-Ch (1000 cases)	0.09	2.61	3.43	1.03	8.70	6.92
Low	0.08	2.26	2.97	0.89	7.54	6.00
High	0.1	3.1	4.1	1.2	10.4	8.3
CRS-Ad (1000 cases)	0.20	5.98	7.85	2.36	19.92	15.84
Low	0.17	5.09	6.68	2.00	16.95	13.48
High	0.23	6.87	9.02	2.71	22.89	18.20
AA (mill. cases)	0.01	0.31	0.40	0.12	1.03	0.82
Low	0.01	0.17	0.23	0.07	0.57	0.46
High	0.03	1.02	1.34	0.40	3.39	2.70