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Kristin Aunan
Jinghua Fang
Guanghai Li
Hans Martin Seip
Haakon Vennemo

Universitetet i Oslo
University of Oslo

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Kristin Aunan*
Jinghua Fang#
Guanghai Li§
Hans Martin Seip*
Haakon Vennemo$

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* CICERO – Center for International Climate and Environmental Research-Oslo
# Taiyuan University of Technology
§ MIT (Massachusetts Institute of Technology) - Center for International Studies
$ ECON – Center for Economic Analyses

CICERO
Center for International Climate
and Environmental Research
P.B. 1129 Blindern
N-0318 Oslo, Norway
Phone: +47 22 85 87 50
Fax: +47 22 85 87 51
E-mail: admin@cicero.uio.no
Web: www.cicero.uio.no

CICERO Senter for klimaforskning
P.B. 1129 Blindern, 0318 Oslo
Telefon: 22 85 87 50
Faks: 22 85 87 51
E-post: admin@cicero.uio.no
Nett: www.cicero.uio.no

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 epidemiology, 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. Estimates of the present agricultural crop loss attributable to enhanced levels of surface ozone are also given. The impact of emission reductions within Shanxi province is, however, limited due to the regional feature of ozone formation. This first assessment of CO_2-reducing abatement options in Shanxi demonstrates that the measures are profitable in a socioeconomic sense. However, the study also demonstrates a certain lack of synergy between the options with respect to their effectiveness in meeting local, regional and global environmental concerns.

Contents

1 Introduction

Emissions of greenhouse gases (GHG) increase the risk of more rapid climate changes than have ever occurred earlier in the history of mankind. Climate changes may have consequences that will severely affect human health directly and indirectly (Hulme et al., 1999). Several model studies indicate that reducing the emissions of GHG in a long-term perspective may entail substantial benefits for instance by reducing the risk of extreme weather events, the risk of increased transmission of malaria and other vector borne diseases and the risk of increased sea level. The impact of such analyses on policy making may, however, be weak due to the large uncertainties and the long-term perspective. Much of the debate related to climate policy has its root in the dilemma that the potential benefits of present mitigating efforts are gained by future generations. Traditional methods for calculating future economic benefits drastically reduce benefits related to climate policy occurring far into the future, hence the conclusion easily drawn is that it is economically rational to postpone action (e.g. Nordhaus, 1993).

Model studies indicate that the effects of climate changes, as drought, sea level rise, and entailed effects related to agricultural production and population migration, are likely to be more severe in developing countries (Martens et al., 1999 and Nicholls et al., 1999). The threshold for effects of climate change having devastating consequences is likely to be lower in these countries, due to a lower adaptation capacity (Handmer et al., 1999). Even though climate scenarios also indicate severe effects in the rich parts of the world, the impacts in terms of human suffering and economic loss may be moderated by a higher socioeconomic capacity of implementing adaptation measures.

Until recently much of the debate on climate policy and the costs of reducing greenhouse gases has not taken into account the possibly large co-benefits that GHG emission reductions may have. Co-benefits (also called secondary or ancillary) benefits are for instance reduced damage to human health and environment from air pollutants, and these benefits appear within a relatively short time horizon after the emissions are reduced. Important sources of GHG, as power production and the transport sector, are also main sources of particulates, SO_2, NO_x, and various hydrocarbons. In addition to the effects of these primary components, there are severe problems related to components produced secondarily in the atmosphere, e.g. ozone and fine secondary particulates.

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 the concentrations of SO_2 and particulates in air violate the WHO air quality guidelines severalfold in many cities. Because typical “1. generation” outdoor 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).

Whereas the climate change issue may not be given high priority on the political agenda, there is in many developing countries 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” aiming at having most big cities satisfying the national 2nd class standards within 3 – 5 years. Regulations of the use of leaded gasoline in major cities and the requirement that all car manufacturers in China should install catalytic converters and electric ignition systems in new cars clearly indicate a policy shift (Gan, 1999).

There are, however, not necessarily large synergies between local and global environmental targets. A study from Mexico City, for instance, showed that an abatement plan that would reduce the emissions of local air pollutants by 64% (via cleaner technology and fuel) would reduce the GHG emissions by only 7%. In Santiago in Chile vehicle emissions standards that would reduce local pollution with 65% were shown to entail a GHG emission reduction of only 5% (Eskeland and Xie, 1998). A study at IIASA (International Institute for Applied Systems Analysis) demonstrates that policies pursuing *either* local pollution targets (with respect to sulfur emissions) *or* global targets (with respect to CO_2 emissions) isolated, without considering the cross-linkages, may have unfortunate effects and perhaps even result in some regions of the world being worse off than if there were no such policies at all (McDonald, 1999). An abatement strategy that is cost-effective with respect to a global perspective may thus not be cost-effective with respect to a local or regional perspective. Generally speaking, the measures that have the largest potentials for synergies are those that leads 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. Traditional abatement measures devised specifically towards reducing SO_2 and the coarser fractions of particulates (> ~ 1µm in diameter), i.e. typical “1. generation measures”, seldom have a significant impact on GHG emissions.

In this study the primary perspective is the global, and we have sought to identify viable options for reducing emissions of GHG in the Shanxi province in China, a province relying heavily on coal in its energy sector. By estimating the local and regional air pollution effects of implementing abatement measures, we aim at sorting out options that have as high as possible synergies, primarily in terms of reducing damage to human health, but also with respect to agricultural production, building materials and cultural heritage. In the first phase, documented in this report, 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.

2 Features of Shanxi

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). Generally, the central axis of the province has the highest population density, the most populated counties being Taiyuan and Datong (see maps in Annex 1).

Shanxi has 2.13 million hectares of forest, which cover 13.8 percent of the province's land surface area (figures for 1990). The tree species include mainly chinese pine, oak, poplar, birch and dragon spruce. At the Loess Plateau in western Shanxi the vegetation is sparse. Shanxi soils are not considered sensitive to acid deposition. Possible effects on vegetation of SO_2, the main precursor of acid precipitation, must therefore be direct, not through soils.

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 (Datong,Yangquan, Xishan, Fenxi, Lu'an, Jincheng, Xuanggang and Huoxian) and 3,000 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 world 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.

Water shortage and water pollution are major problems in Shanxi. Concerning water pollution, run-off from agriculture and entailed eutrophication represents a large problem (indicated by high levels of ammonia nitrogen and COD (carbon oxygen demand) and BOD (biologic oxygen demand). In addition, high levels of volatile phenol, petroleum and suspended matters are reported. Groundwater quality is a problem in several cities (Wu, 1997). A large project is under construction to divert water from the Yellow River to the province to alleviate the water shortage. In this connection a hydropower station will also be built. With a planned installed capacity of 1.08 GW it will play an important role in the North China grid whose main supply at present is from coal fired thermal power plants.

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).

3 Air pollution in Shanxi cities

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 amount to 7.1% of the production (National Bureau of Statistics, 1999). The country has been the world's largest coal producer and consumer since 1988 and with an estimated 11.6% of the world's coal reserves, 2.3% of oil reserves and 0.9% of gas reserves is likely to depend heavily on coal into the future (BP Amoco, 2000).

Table 1 shows the annual coal consumption in Shanxi (average for the period 1995-1997). The largest single consumer is the coke making sector. A large share of the coke is exported to other provinces in China. Based on the figures in Table 1, we have estimated the total emission of TSP and SO_2 from coal consumption in Shanxi (see Table 2). We have assumed uniform emission factors in the different source groups, 20.2 ktons SO_2 and 11.3 ktons of TSP per mill. tons of raw coal, except for coke making, where we assumed a lower emission factor for SO_2. This adjustment was done because we had to take into consideration that much of the sulfur remains in the coke during the coking and much of the coke is exported. As a preliminary estimate we assumed an emission factor of 10% of the one used for other sources. This may be too low and more accurate information is needed.

Table 1. Annual coal consumption in Shanxi, average for 1995-1997 (mill. tons of raw coal).

	Mill. tons	% of total
Total	136.15	100.0
Households	5.88	4.3
Rural	0.97	0.7
Urban	4.92	3.6
Industry	130.26	95.7
Power	28.33	20.8
Coke making	72.06	52.9
Industrial combustion	29.87	21.9

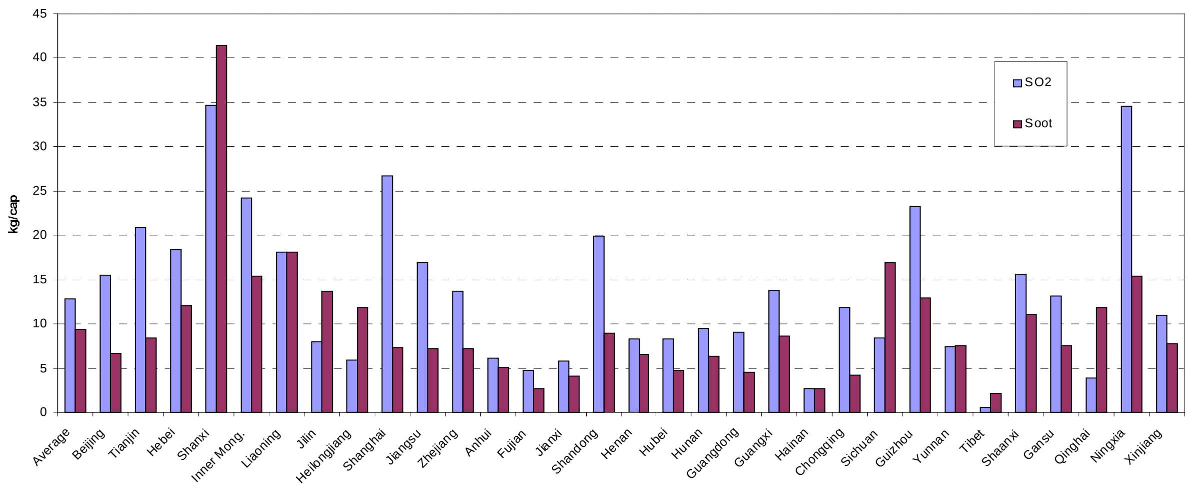
Table 2. Annual estimated emissions of TSP and SO_2 from coal consumption in Shanxi, average for 1995-1997).

	TSP (1000 tons)	SO_2 (1000 tons)
Total	1531.7	1437.2*
Households	66.2	118.6
Rural	10.9	19.5
Urban	55.3	99.1
Industry	1465.5	1318.6
Power	318.8	571.2
Coke making	810.7	145.3
Industrial combustion	336.0	602.1

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

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 latter is partly indicated by the fact that the per capita SO_2 emission from *non-industrial* sources in Shanxi is 2.5 times the country average (about 10 kg/cap), exceeded only by Chongqing and Guizhou. The total per capita emissions of SO_2 (all sources) and soot (from industry) are the highest in China, and is for soot 4.4 times the country average (see Figure 1). The waste gas soot cleaning ratio in the industry is among the lowest in China (only Tibet, which has low emissions, has a lower ratio) (National Bureau of Statistics, 1999).

Figure 1. Annual emissions of SO_2 and soot (industrial sources) per capita (kg/cap) in China, by province, 1998. (National Bureau of Statistics, 1999).



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 µg/m^3 as an annual average (see Figure 2). 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 µg/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). As seen from Figure 2 these thresholds are exceeded severalfold in many cities in Shanxi. 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 µg/m^3 (annual average NO_2 level) is likely not exceeded in many cities.

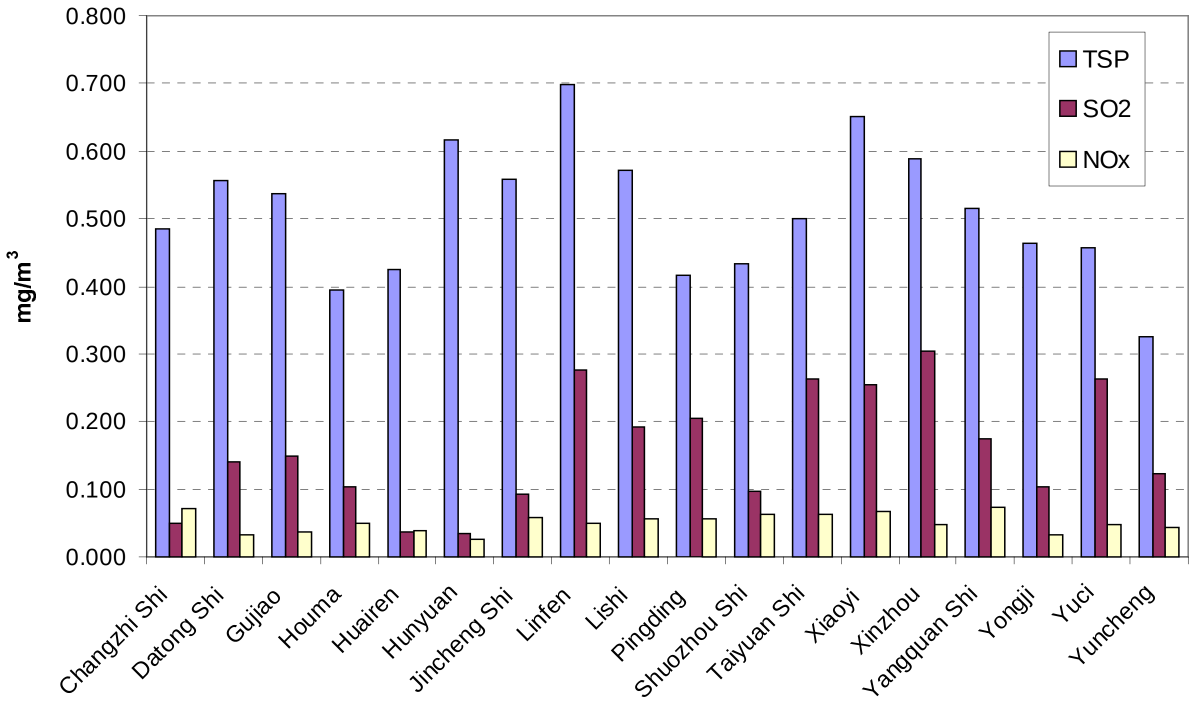
The province capital *Taiyuan* is situated in a mountain basin in central Shanxi, surrounded by hills and mountains on three sides. The topography leads to periods of inversion and stagnant air masses during winter time, thus enhancing the concentration of air pollutants. The main emission sources in 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.

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 $\mu g/m^3$.

Figure 2. Air quality in some cities in Shanxi (annual average for 1997 and 1998). For location of the cities, see map in Annex 1). Source: Shanxi EPB, 1999.



4 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 3. 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. 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. Abatement cost per CO_2 unit and the reduction potentials for various components are rendered in Table 5. The assumed emission reduction potentials, as share of the baseline emission coefficient, are given in Table 4. Assumed emission reduction potential of the abatement options (as percentage of baseline emission coefficient). In the following calculations we have considered each measure in isolation. 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 3. Potentials for the abatement options. Amount (mill. tons of raw coal consumed) and percentage share of source consumption eligible to the selected abatement options.

	Rural households	Power	Industrial combustion	SUM mill. tons	SUM % of tot.
Coal washing (mill. tons)		23.8	29.9	53.7	
% of source		84.1	100.0		
% of total		17.5	21.9		39.4
Briquetting (mill. tons)	1.0		29.9	30.8	
% of source	100.0		100.0		
% of total	0.7		21.9		22.6
Improved management (mill. tons)			22.4	22.4	
% of source			75.0		
% of total			16.5		16.5
Boiler replacement (mill. tons)			22.4	22.4	
% of source			75.0		
% of total			16.5		16.5
Co-generation (mill. tons)[1]			0.7	0.7	
% of source			2.3		
% of total			0.5		0.5
Modified boiler design (mill. tons)[2]		17.0	17.9	34.9	
% of source		60.0	60.0		
% of total		12.5	13.2		25.6

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

Table 4. Assumed emission reduction potential of the abatement options (as percentage of baseline emission coefficient).

	CO_2	TSP	SO_2
Cogeneration	21	21	21
Modified boiler design	17	17	17
Boiler replacement	25	25	25
Improved boiler management	8	8	8
Coal washing	10	15	40
Briquetting	10	45	35*

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

Table 5. 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	-30.0	0.3	1.6	2.9
Modified boiler design	-6.3	12.8	65.5	117.3
Boiler replacement	-2.6	12.3	63.0	112.9
Improved boiler management	9.2	3.7	18.9	33.9
Coal washing	23.9	11.8	90.6	433.0
Briquetting	26.6	6.8	156.1	217.6

5 Health damage from outdoor air pollution in Shanxi cities

5.1 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. These functions are in the following used to assess the possible health impact that outdoor air pollution may have at present in some cities in Shanxi. In addition to the functions from Chinese studies, we also rely on some studies from Europe and the USA. 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 recommended 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 increase in number of cases per mill. inhabitants per $\mu g/m^3$. In other words, this is a way of 'calibrating' the functions according to given conditions.

As several of the health indicators necessary to calibrate the exposure-response functions for Shanxi have not been available in this study, we decided to rely on data from a study in Guangzhou in the Guangdong province (Aunan and Li, 1999). The functions, shown in Table 6, 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 average length of this period is indicated in Table 6. The background for the functions and baseline frequencies used in the calibration is given in Annex 2.

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; see Annex 2).

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 the areas where most epidemiological studies have taken place. 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 is 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., 2000 and reference therein).

Table 6. Exposure-response functions used in the calculation of health effects of air pollution. Change in annual number of cases per million people (all ages) per 1 $\mu g/m^3$ change in ambient concentration[1]. Uncertainty intervals represent tentatively ±1SD. (See Annex 2 for a description of how the functions are derived).

End-point	Indicator component	Period per case	Coefficient (uncertainty interval)
Premature deaths (adults)	PM_{10}		2.2 (0-4.1)
	SO_2		4 (3-6)
Infant deaths	PM_{10}		1.2 (0.8-1.7)
	SO_2		0.3 (-0.4-1.1)
Outpatient visits (OPV)	PM_{10}		4,670 (1,980-7,360)
	SO_2		1,800 (1,510-2,100)
Emergency room visits (ERV)	PM_{10}		55 (15-95)
	SO_2		186 (112-260)
Hospital admissions (HA)	PM_{10}	21 days	97 (65-121)
	SO_2	"	186 (89-302)
Work day loss (WDL)	PM_{10}	1 day	18,400 (9,200-27,600)
Respiratory symptoms in children (ARS-Ch)	PM_{10}	"	21,500 (14,190-32,470)
	SO_2	"	2,830 (2,690-2,970)
Respiratory symptoms in adults (ARS-Ad)	PM_{10}	"	28,320 (21,130-35,520)
	SO_2	"	7,650 (7,270-8,030)
Chronic resp. sympt. in children (CRS-Ch)	PM_{10}	~1 year	15 (13-18)
Chronic resp. sympt. in adults (CRS-Ad)	PM_{10}	"	34 (29-39)
Asthma attacks	PM_{10}	1 day	1,770 (990-5,850)

[1] The share of the population that is <14 y, 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.

Use of unvented coal stoves is still common in Shanxi and causes 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). 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. A study by Wang and Smith (1999a, b), for instance, showed that the health benefit was much higher per ton reduction in particulates emissions from household stoves than from coal-fired power plants.

5.2 Estimated health impacts and possible reductions from abatement measures

In order to make a rough estimate of the impacts of outdoor air pollution on human health in Shanxi, we may assume that the share of the population that is exposed to health damaging levels mainly live in cities and townships. We therefore applied the estimated population weighted exposure (PWE[1]) in the 18 cities from which we had pollution data, on the total urban population in Shanxi, which constitutes 51% of the total population (1990 figures, CIESIN/SEDAC, 1997[2]). This is a rather conservative approach, for two reasons. Firstly, the way we estimated the *urban* population using official statistics, may lead to an underestimate of the actual share that lives in non-rural areas (cities and townships), due to urbanisation and agglomeration of towns and villages not accounted for in the census data. Secondly, due to the widespread use of coal for multiple purposes in Shanxi, the level of air pollution is likely high enough to cause health damage also in villages and rural areas, as indicated for instance by the study by Salmon et al. (1995) in the small village of Yungang.

On the other hand, we do not know whether the air pollution situation in the 18 cities from which we have data, might be higher than average for all cities in the province. If it is, applying the population weighted exposure from these cities to the total urban population, may overstate the impact.

Table 7 shows the estimated number of excess cases given the present level of air pollution (0% reduction). These estimates only give a rough indication of the possible benefits that could be obtained if the pollution level was reduced. Because the functions are linearised and in the original study only applies to a limited marginal change in pollution level, often a few tens of $\mu g/m^3$, there are very large uncertainties in estimating excess cases in this way. More relevant is perhaps to look at the possible benefits that may be obtained by a moderate reduction in pollution level, say 5-10%. Scenarios where the average level of particulate air pollution is reduced by a flat percentage across the urban areas (5 and 10% reduction) are therefore also shown in Table 7.

The population weighted PM_{10} exposure in the 18 cities is 280 $\mu g/m^3$. The corresponding level for SO_2 is 186 $\mu g/m^3$. In the calculation of weighted exposure we assumed that the

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[1] PWE for a given pollutant was calculated as: $\frac{1}{P}\sum_i c_i \cdot p_i$, 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.

[2] The urban population is estimated as being the total population minus the rural population living outside townships (*zhen*) (i.e. in villages (*xiang*)).

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).

As described in Annex 2, estimated health effects from using the exposure-response functions for PM_{10} and SO_2 cannot be added. The estimated health damage obtained by using the functions having PM_{10} as independent variable is usually higher than for the SO_2 functions. An exception is emergency room visits (ERV), where the estimate gets twice as high with the SO_2 function. The estimated effect on hospital admissions (HA) and mortality (applying the function from European studies) also gets somewhat higher when applying SO_2 functions.

According to the estimate obtained by the PM_{10} function, about 10% of the annual deaths in urban Shanxi are premature due to air pollution (11% when using the SO_2 function)[3]. Other studies in Central and Eastern Europe (CEE) have estimated an attributable risk of air pollution to excess annual mortality rate of 3% (World Bank, 1993) and 6% (Aunan et al., 1998). Premature deaths due to pollution exposure are related to certain types of illnesses and fall disproportionately on the elderly and those with an already compromised health. The degree of prematurity (lost life years) is not known from the Chinese study being the basis for the function applied here, nor for the studies in western countries, used in the studies in CEE.

Table 7. Estimated excess cases attributable to outdoor particulate air pollution in urban Shanxi, at present level (0% reductions) and at a 5 and 10% reduction in the concentration level. See Table 6 for abbreviations .

% reduction	Premature deaths (adults)	Infant deaths	OPV (mill. cases)	ERV (1000 cases)	HA (1000 cases)	WDL (mill.)	ARS-Ch (mill. cases)	ARS-Ad (mill. cases)	CRS-Ad (1000 cases)	AA (mill. cases)
0	9992	5418	21.2	249.8	440.5	83.6	97.7	128.6	156.0	8.0
5	9617	5215	20.4	240.4	424.0	80.4	94.0	123.8	151.3	7.7
10	9242	5012	19.6	231.1	407.5	77.3	90.3	119.0	145.5	7.4

How implementation of the abatement options may reduce the average population exposure level in urban areas was estimated by relying on several assumptions. First, we assumed that the concentration data that we had were representative for the cities and we assumed 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.

Secondly, we assumed proportionality between the emission of the respective component from coal consumption in each source category (see Table 2) and their contribution to

[3] The annual death rate in Shanxi is 6.17 per 1000 (1998) (National Bureau of Statistics, 1999)

the annual PWE, while at the same time assuming that 25% of the estimated PWE for TSP and 15% of the estimated PWE for SO_2 would not be influenced by the abatement options. In lack of more detailed information at this stage, this is in our view a reasonably satisfying assumption, 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 Spadare, 1999). From this perspective the source contribution to the long-term average concentrations in urban areas (and subsequently PWE) would differ from the source distribution of coal consumption only to the extent that other sources are contributing and to the extent that the level is influenced by neighbouring provinces (the magnitude of the regional background level). 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, and a ratio of regional background plus contribution from other sources versus contribution from coal consumption of 15:100 seems reasonable (15% of PWE is slightly lower that the lowest SO_2 level recorded in any of the cities). Concerning particulate air pollution 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 may, until more data become available, 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). Concerning the impact of other sources than coal consumption, we have used data for source contribution to particulate air pollution in Guangzhou (Aarhus et al., 1999) as a basis for preliminary estimating a ratio of regional background plus contribution from other sources versus contribution from coal consumption of 25:100.

By combining the information in Table 2 and Table 3, we may obtain a rough estimate of the relative potential for reducing outdoor PWE for particles and SO_2 for each abatement option:

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

where:

$\Delta PWE_{i,}$	=	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[4])
S_k	=	relative contribution of <u>s</u>ource *k* eligible to option *j* to PWE (based on Table 2)
$E_{j,k}$	=	share of source *k* <u>e</u>ligible to option *j* (see Table 3)
R_j	=	<u>r</u>eduction potential of abatement option *j* (see Table 4)

The calculated ΔPWE for particles and SO_2 are given in Table 8. The estimates apply to each abatement option implemented separately. Since some of them relate to the same

[4] Thus, PWE is 158 μg/m^3 for SO_2 and 210 μg/m^3 for PM_{10}.

source group, the effect of one option may be reduced if another option has already been implemented. We have assumed a PM_{10}/TSP conversion rate of 0.55 (EPA, 1982). As noted in Annex 2 there are some indications that this may be somewhat high as a general assumption in China, but no data are to our knowledge available to suggest another factor. The corresponding ratio suggested for $PM_{2.5}/PM_{10}$ is ~ 0.6 (Dockery and Pope, 1994), hence the $PM_{2.5}$/TSP ratio should be about 0.33. In light of the measurements from Yungang, mentioned above, where the $PM_{2.1}$/TSP ratio was 0.26 (Salmon et al., 1995) a PM_{10}/TSP ratio of 0.55 seems to be a reasonable assumption until more data become available.

Table 8. Estimated reduced population weighted exposure (ΔPWE) for PM_{10} and SO_2 in urban Shanxi from implementing abatement measures (μg/m³).

	ΔPWE (μg/m³)	
	PM_{10}	SO_2
Cogeneration	0.2	0.3
Modified boiler design	9.0	12.9
Boiler replacement	8.6	12.4
Improved boiler management	2.6	3.7
Coal washing	12.4	47.7
Briquetting	21.4	24.0*

*See footnote to Table 4.

The health effects, in terms of the different end-points for which we had exposure-response functions (see Table 6), were calculated for each abatement option as implemented separately by applying the estimated ΔPWE for PM_{10}, see Table 9.

In a case study on the power and household energy sectors in China, Wang and Smith (1999a, b) estimated that the health effects of reducing emissions from household stoves were about 40 times higher than the effects of reducing emissions from power plants. This was based on a careful analysis of population exposure and to what extent emissions from the two source categories contribute to the dose actually inhaled by different population groups (i.a. a stack height plus plume rise of 75 meters was assumed). They arrived at an estimated health impact, in terms of premature mortality, of 0.002 deaths per ton PM_{10} emitted from power plants, and correspondingly, 0.08 deaths per ton PM_{10} emitted from household stoves. Our study gives 0.005 deaths per ton TSP reduction (all source categories), and thus, in the light of the simplified methodology which addresses the impact in urban areas of all source groups, and which only apply to outdoor air pollution, is in line with the estimate by Wang and Smith.

As seen from Table 5 and Table 9, briquetting, which has the largest potential of reducing health damage, also is the most expensive of the six options, in terms of cost per ton CO_2. This demonstrates that the ranking of options may get drastically changed when going from a mere global perspective (cost per CO_2-reduction) to a local. The economic valuation of the reduced health damage thus becomes crucial to the conclusion, when the aim is to rank the options according to both perspectives.

Table 9. Estimated health effects 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)	8	320	308	92	443	764
Low	0	0	0	0	0	0
High	15	597	574	172	826	1423
Infant mortality	4	174	167	50	240	414
Low	3	109	105	31	151	260
High	6	238	229	69	330	568
OPV (mill. cases)	0.02	0.68	0.65	0.20	0.94	1.62
Low	0.01	0.29	0.28	0.08	0.40	0.69
High	0.03	1.07	1.03	0.31	1.48	2.55
ERV (1000 cases)	0.20	8.01	7.71	2.31	11.08	19.09
Low	0.05	2.18	2.10	0.63	3.02	5.21
High	0.34	13.83	13.31	3.99	19.14	32.98
HA (1000 cases)	0.35	14.12	13.59	4.08	19.55	33.67
Low	0.24	9.46	9.11	2.73	13.10	22.56
High	0.44	17.62	16.95	5.09	24.38	42.00
WDL (mill.)	0.07	2.68	2.58	0.77	3.71	6.39
Low	0.03	1.34	1.29	0.39	1.85	3.19
High	0.10	4.02	3.87	1.16	5.56	9.58
ARS-Ch (mill. cases)	0.08	3.13	3.01	0.90	4.33	7.46
Low	0.05	2.07	1.99	0.60	2.86	4.93
High	0.12	4.73	4.55	1.36	6.54	11.27
ARS-Ad (mill. cases)	0.10	4.12	3.97	1.19	5.71	9.83
Low	0.08	3.08	2.96	0.89	4.26	7.34
High	0.13	5.17	4.98	1.49	7.16	12.33
CRS-Ad (1000 cases)	0.12	5.00	4.81	1.44	6.92	11.92
Low	0.11	4.26	4.10	1.23	5.89	10.15
High	0.14	5.75	5.53	1.66	7.95	13.70
AA (mill. cases)	0.01	0.26	0.25	0.07	0.36	0.61
Low	0.00	0.14	0.14	0.04	0.20	0.34
High	0.02	0.85	0.82	0.25	1.18	2.03

Figure 3 shows the estimated reduced PWE (ΔPWE) in urban Shanxi per mill. ton CO_2 reduction for all abatement options, when implemented separately. As seen, the relationship between these two variables is similar for all options that relate to improving energy efficiency in the combustion, whereas treatment of the coal before burning, i.e. briquetting and coal washing, have a considerably higher ratio, simply because of the lower reduction of CO_2 compared to SO_2 and PM_{10} (see Table 4). This implies that for typical energy efficiency options the cost-efficiency ranking in terms of mill. USD/ΔPWE (PM_{10} or SO_2) concurs with the ranking according to USD/ton CO_2, whereas for the other types of options the ranking differs (this is shown for ΔPWE_{PM10} and CO_2 in Figure 4.

Figure 3. Reduced population weighted exposure (μg/m³) per mill. ton CO_2 reduction for the abatement options, when implemented separately and for the total estimated potential in Shanxi.

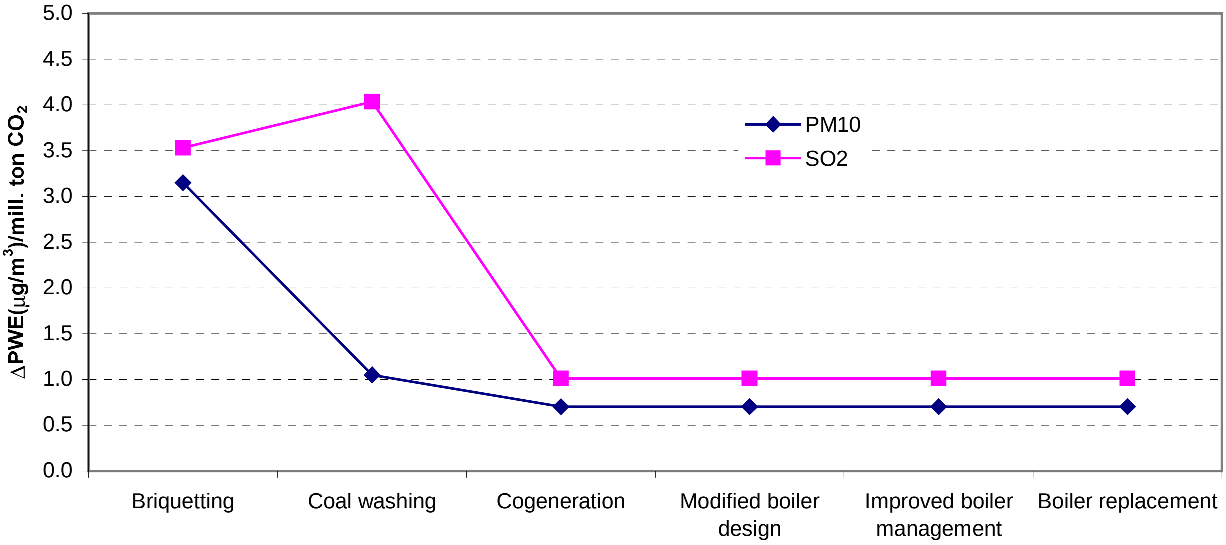
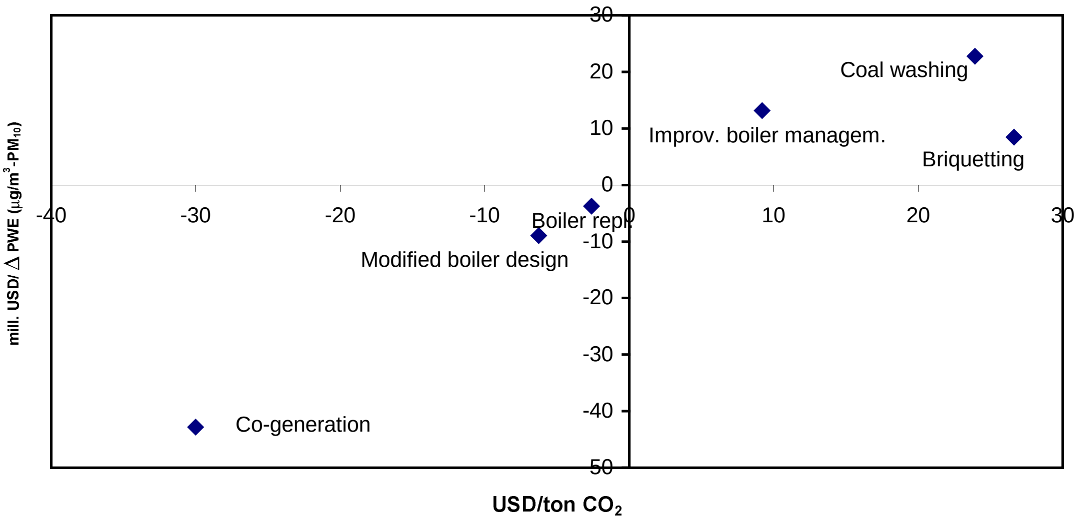


Figure 4. Cost efficiency of abatement options in terms of USD/ton CO_2 (x-axis) versus mill. USD/ΔPWE (μg/m³ PM_{10}) (y-axis).



6 Valuing health benefits

6.1 Assumptions of unit prices

Although PWE and estimates of cases of different health effects attributable to air pollution are useful statistics of health impacts of local and regional pollution, it is sometimes advantageous to include an explicit monetary value of 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 by which to assess investments in abatement and energy conservation. This section provides some tentative estimates of the value of the reduced health impacts of local and regional pollution.

The gain that an explicit valuation of pollution brings, comes at the cost of adding more assumptions. Assumption number one is that the health impacts are as summarized in Table 9. That is, we continue to build on the effect chain of impacts that has been developed in this paper. Assumption number two concerns the unit prices to put on each impact of table 9. The unit prices we employ are summarized in Table 10.

Table 10. Unit prices of health impacts (end-points).

Impact (End-point)	Unit price ($)	Remark
Premature deaths (adults)	63 000	100*GDP/capita Shanxi.
Infant deaths	63 000	
Outpatient visits	12.5	Assuming 100 RMB
Emergency room visits	12.5	Assuming 100 RMB
Hospital admissions	5123	Source Aarhus et.al. (2000)
Work day loss	1.75	GDP/(capita*day) in Shanxi
Acute respiratory symptoms in children	12.5	Assuming 100 RMB
Acute respiratory symptoms in adults	12.5	Assuming 100 RMB
Chronic respiratory symptoms in children	4 700	Modified WTP estimate is (1990) $175 000 in U.S. Multiplied by relative GDP/capita in Shanxi (1998) to U.S. (1990)
Chronic respiratory symptoms in adults	4 700	Modified WTP estimate is (1990) $175 000 in U.S. Multiplied by relative GDP/capita in Shanxi (1998) to U.S. (1990)
Asthma attacks	1	WTP estimate is $32 in U.S. 1990. Multiplied by relative GDP/capita in Shanxi (1998) to U.S. (1990)

Note: GDP/capita (1998) = 5040 RMB in Shanxi, which equals $630 (National Bureau of Statistics, 1999). GDP/capita (1990) in U.S. was $23 200 (U.S. Department of Commerce, Bureau of Economic Analysis).

It is of course difficult to estimate a unit price of health end-points. The estimates are uncertain, and one should indicate the uncertainty. We chose in this application not to include explicit subjective confidence intervals around the estimates, but emphasise that our estimates have an illustrative ring to them and indicate the main features of the data.

In order to indicate the main features of the data we do not distinguish between the importance of effects on children versus adults. We assume that acute respiratory symptoms, hospital admissions and outpatient visits can be treated more or less the same by an estimate of 100 RMB per case. 100 RMB is a reasonable estimate of medicine (approximately 50 RMB), other expenses including round trip taxi and admission fee (approximately 25 RMB) and addition of some willingness to pay. Western estimates of these symptoms go from $18 for a case of acute respiratory symptoms for children (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). Our estimate of a work day loss simply equals GDP/capita in Shanxi.

The serious impacts, premature deaths and chronic respiratory symptoms, have higher unit prices that give them higher weights in the final evaluation. To arrive at our preferred estimate of premature deaths we have done the following: First we have consulted Western studies and surveys of the so called value of a statistical life (VSL). A value of a statistical life is actually a value of a small change in risk. Three estimates are the one 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 U.S. The mean of the distribution is used in their analysis, $ 4.8 million. 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. Table 11 reports the shares of GDP/capita that these estimates amount to.

Table 11. The value of risk reduction in different countries, given as the ratio of VSL/GDP/capita.

Country	U.S.	E.U.	Norway
Ratio	206	155	70

Note: The U.S. data (1990) uses VSL $4.8 million and GDP/capita of $23 200. The E.U. data (1995) uses VSL of $3.1 million and GDP/capita of $20 000. The Norwegian data (1999) uses VSL of $2 million and GDP/capita of $28 400.

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 U.S. 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 U.S. 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 U.S to account for differences in standards of living and thus in willingness (or ability) to pay.

6.2 Environmental costs

Table 12 summarises the environmental cost that we estimate.

Table 12. Estimated costs of health effects from reduced emission of particles obtainable from implementing the abatement options separately (mill. USD). 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	Bri-uetting
Mortality (M)	0.50	20.16	19.40	5.80	27.91	48.13
Low	0.00	0.00	0.00	0.00	0.00	0.00
High	0.95	37.61	36.16	10.84	52.04	89.65
Infant mortality	0.25	10.96	10.52	3.15	15.12	26.08
Low	0.19	6.87	6.62	1.95	9.51	16.38
High	0.38	14.99	14.43	4.35	20.79	35.78
OPV	0.25	8.50	8.13	2.50	11.75	20.25
Low	0.13	3.63	3.50	1.00	5.00	8.63
High	0.38	13.38	12.88	3.88	18.50	31.88
ERV	0.00	0.10	0.10	0.03	0.14	0.24
Low	0.00	0.03	0.03	0.01	0.04	0.07
High	0.00	0.17	0.17	0.05	0.24	0.41
HA	1.79	72.34	69.62	20.90	100.15	172.49
Low	1.23	48.46	46.67	13.99	67.11	115.57
High	2.25	90.27	86.83	26.08	124.90	215.17
WDL	0.12	4.69	4.52	1.35	6.49	11.18
Low	0.05	2.35	2.26	0.68	3.24	5.58
High	0.18	7.04	6.77	2.03	9.73	16.77
ARS-Ch	1.00	39.13	37.63	11.25	54.13	93.25
Low	0.63	25.88	24.88	7.50	35.75	61.63
High	1.50	59.13	56.88	17.00	81.75	140.88
ARS-Ad	1.25	51.50	49.63	14.88	71.38	122.88
Low	1.00	38.50	37.00	11.13	53.25	91.75
High	1.63	64.63	62.25	18.63	89.50	154.13
CRS-Ad	0.56	23.50	22.61	6.77	32.52	56.02
Low	0.52	20.02	19.27	5.78	27.68	47.71
High	0.66	27.03	25.99	7.80	37.37	64.39
AA	0.01	0.26	0.25	0.07	0.36	0.61
Low	0.00	0.14	0.14	0.04	0.20	0.34
High	0.02	0.85	0.82	0.25	1.18	2.03
Sum (expected value)	5.75	231.13	222.39	66.69	319.95	551.14

The environmental cost estimates indicate that briquetting is the option with far the highest local environmental benefits. The environmental benefits of briquetting run up to 0.5 *billion* US dollars. Several of the other options also bring benefits larger than 100 million dollars. One cannot add the benefits of the options, as several of them target the same pollution reduction potential.

We have not attempted to quantify the uncertainty of the aggregate benefit estimates, but use a confidence interval on each benefit that corresponds to the interval of the underlying physical end-point.

7 Net cost of CO_2 reductions

This section combines the estimated abatement costs of six options (Table 5) and their associated environmental benefits (Table 12) to estimate the net unit cost of CO_2-reductions. The net cost is the abatement cost less the environmental benefit, per ton of CO_2 removed. The net costs that we find are indicated in Table 13.

Table 13. The net cost of CO_2-reductions.

	Abatement cost (US$/ton CO_2)	Local environmental benefit (US$/ton CO_2)	Net cost of CO_2-reductions (US$/ton CO_2)	Potential CO_2-reduction (mill. tons)
Co-generation	-30.0	19.2	-49.2	0.3
Modified boiler design	-6.3	18.1	-24.4	12.8
Boiler replacement	-2.6	18.1	-20.7	12.3
Improved boiler management	9.2	18.0	-8.8	3.7
Coal washing	23.9	27.1	-3.2	11.8
Briquetting	26.6	81.1	-54.5	6.8

Note: the net cost of CO_2-reductions (column 3) is the abatement cost (column 1) less the environmental benefit (column 2)

Table 13 indicates that after accounting for local environmental benefits, the net cost of each option is negative. To put it more simply, each option is profitable in the social sense. (Three of them, with negative abatement costs, are profitable in a business sense as well).

Although every option carries large environmental benefits on a per ton basis, the briquetting option clearly carries the largest benefit. The benefit makes this option the most profitable in the social sense, although it has the largest abatement cost. The reason why briquetting has such a large environmental benefit, is that in addition to reducing CO_2, it reduces particle and SO_2 emissions to almost half of their former levels (cf. Table 4). However, it must be remembered that this is not true for the types of briquettes commonly used in Shanxi at present.

If we in future work also would be able to add in the benefits of increased agricultural productivity (from less ozone – see below) and less material damage, the CO_2-abatement options probably would be shown to be even more profitable in the social sense. However, recall that our estimates are illustrative. They are the best we can do at present, but there are many sources of uncertainty which need further consideration.

8 Possible present crop losses due to surface ozone

The phytotoxicity of ozone is well documented, and it has been shown that ozone may significantly reduce the yield of several types of agricultural crops (US-EPA, 1996). The possible increase in surface ozone may be a cause of concern regarding the prospects for Chinese agricultural production. In Aunan et al. (2000) it is showed that the yield of the main crops may be substantially reduced in the future due to increased ozone concentrations, and that the yield of spring wheat, soybean and corn may already be affected. In the following we have applied the same methodology as in Aunan et al. (2000) to estimate possible present crop loss due to ozone in Shanxi. The impact of the abatement measures discussed above on the O_3 level in Shanxi has not been estimated, but are likely to be minor, due to the impact of long-range air pollution on the ozone levels.

The main food grains grown in Shanxi are corn, winter wheat, millet, and sorghum. Other important food crops are soybeans and potatoes. The main cash crops of Shanxi include cotton, flaxseed, sugar beets and tobacco. Cotton is grown mainly in the Yuncheng and Linfen regions in the southwestern Shanxi. Flaxseed, the main oil-bearing crop of the province, is grown mainly in the north and in the mountain areas.

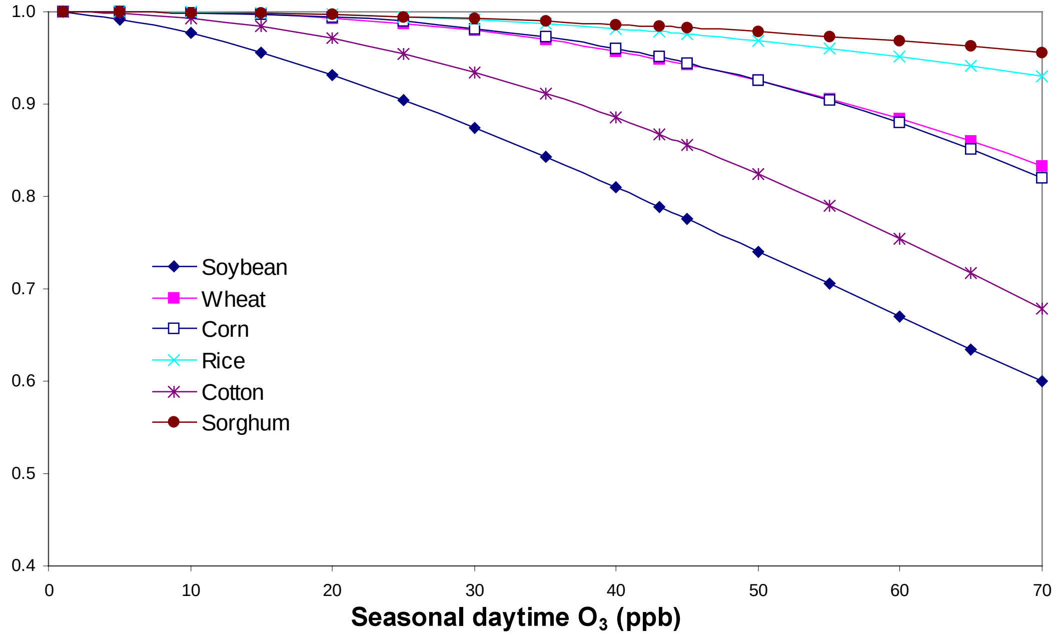
The annual production of those crops for which dose-response function for crop yield and ozone exposure is available is shown in Table 14. In addition, the production of millet, an important crop in Shanxi, is given. Three sets of dose-response functions are applied, see Figure 5. The functions are based on studies in the USA and Europe. Function having seasonal daytime mean (7 or 12 h d^{-1}) O_3 as exposure metric are denoted M7/M12. SUM06 (also denoted 24-h SUM06) is calculated as the sum, over a 3-months' period, of the hourly ozone concentrations for hours when the concentration is at or above 60 ppb. AOT40 is calculated as the sum of differences between the hourly mean concentrations of O_3 and 40 ppb for hours when the concentrations exceeds 40 ppb, for each daylight hour with global radiation ≥50 Wm^{-2} over a 3 months' period.

A global three-dimensional photochemical tracer/transport model (CTM) of the troposphere was used to estimate surface ozone levels, in terms of the three metrics, on a regional scale for China for 1990 and 2020. The scenarios are described in Aunan et al. (2000). The model resolution is 8° latitude and 10° longitude with nine vertical layers below 10hPa (Berntsen and Isaksen, 1997). The approximate average level in Shanxi was estimated by overlaying the model grid and the map with the province borders and weighing the values for the grid cells according to the area of the province included in each grid cell.

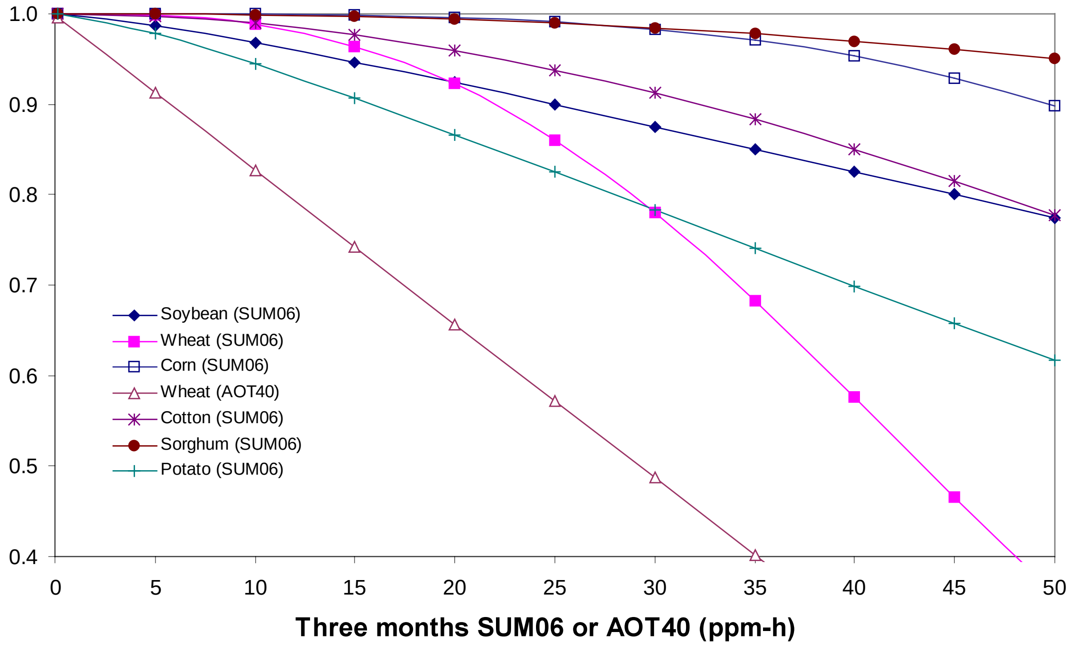
The crop loss estimated by means of the different functions differs substantially. This is mainly due to the different threshold levels for effect that are embedded in the function and to what extent this threshold is exceeded. According to the model, the average O_3 level during daytime is between 40 and 50 ppb during summer months in Shanxi (1990). In the baseline scenario for 2020 (see Aunan, 2000) this level was estimated to reach 63-69 pbb in the summer months, i.e. above the threshold of 60 pbb in the SUM06 functions.

The largest crop losses, in terms of ktons, are estimated for corn and wheat. In terms of relative loss the largest reductions are estimated for soybean and cotton, due to their higher sensitivity to ozone.

Figure 5. Exposure-response functions used in the calculation of crop yield losses. The exposure indices are a) seasonal daytime mean (7 or 12 h d^{-1}) (ppb); b) SUM06 or AOT40 (ppm-h). y-axis shows relative yield.


1.0
0.9
0.8
0.7
0.6
0.5
0.4
0
10
20
30
40
50
60
70
Soybean
Wheat
Corn
Rice
Cotton
Sorghum
Seasonal daytime O3 (ppb)

a)


1.0
0.9
0.8
0.7
0.6
0.5
0.4
0
5
10
15
20
25
30
35
40
45
50
Soybean (SUM06)
Wheat (SUM06)
Corn (SUM06)
Wheat (AOT40)
Cotton (SUM06)
Sorghum (SUM06)
Potato (SUM06)
Three months SUM06 or AOT40 (ppm-h)

b)

Table 14. Annual production of main crops in Shanxi. Average for 1991-1995.

	ktons	% of Chinese prod.
Single crop rice	47	0.0
Spring wheat	101	0.8
Winter wheat	2850	3.2
Soybean	247	1.9
Corn	2950	3.0
Sorghum	559	10.7
Millet	642	18.4
Cotton	129	2.2
Tuber	677	2.3

Table 15. Crop losses due to present levels of surface ozone estimate by means of different dose-response functions (M7/M12; SUM06 and AOT40, see text). HP: Hypothetical crop production estimated for a situation when ozone does not exceed certain threshold levels. (Based on Aunan et al., 2000).

	M7/M12		SUM06		AOT40	
	ktons	% of HP	ktons	% of HP	ktons	% of HP
Single crop rice	0.9	1.8				
Spring wheat	4.51	4.3	0.0	0.02	17.4	14.74
Winter wheat	45.00	1.6	0.0	0.00	41.4	1.45
Soybean	16.82	13.4	2.7	1.07		
Corn	98.63	3.3	0.0	0.00		
Sorghum	6.78	1.2	0.1	0.02		
Cotton	14.3	9.6	0.2	0.2		
Tuber			12.3	1.8		

9 Some preliminary conclusions and needs for further work

This first assessment of CO_2-reducing abatement options within the coal sector in Shanxi demonstrates that the measures are profitable in a socioeconomic sense. The study further indicates a certain lack of synergy between the options with respect to local and global environmental concerns. As seen from Table 5 and Table 9, briquetting and coal washing, which have the largest potentials for reducing health damage, also are the most expensive of the six options in terms of costs per ton CO_2 (not including environmental effects). For those options that do not have a negative cost, the cost-efficiency ranking gets altered when looking at CO_2, TSP and SO_2, respectively (see Figure 4).

This demonstrates that the ranking of options in this case depends on whether a global or local perspective is the primary one. The economic valuation of the benefit thus becomes crucial to the conclusion, when the aim is to outline a policy that takes into consideration both perspectives. A main task in further work will be to further improve the physical and economic assessment of the co-benefits that may be achieved from CO_2-abatement measures in the coal sector in Shanxi. We also need to identify abatement options within the two important source categories coke making and urban households. Additionally, there is a need to identify possible factors that may limit the feasibility of the options, as for instance the impact of water shortage and water pollution on the potential for coal washing.

Whereas we in this study applied similar emission factors for nearly all source categories, this may not be warranted and needs to be investigated in further work, i.a. we need to know more precisely how much of the sulfur in coal remains after coking. Also, a simplified approach was taken to estimate population exposure, which levelled out the large differences between the various cities. There is a need to identify the source contribution to air pollution levels in a more detailed geographical scale, and subsequently where 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 order to calibrate the dose-response functions for health effect, more information is needed about the present health status and use of medical services in Shanxi.

In this report we have limited the focus to health effects of particles and SO_2 and crop loss due to ozone. For the latter we did, however, not estimate the impact of the abatement measures, because this would necessitate more information about NO_x and nmVOC reductions obtainable from the options, in addition to regional modelling of ozone. Concerning material damage the SO_2 concentrations are certainly high enough to cause increased corrosion rates. Whereas recently proposed functions for many materials have SO_2 and O_3 as independent variables, functions employing only SO_2 should probably be used in Shanxi, due to lack of sufficient O_3-data (Kucera and Fitz, 1995; and Henriksen and Hågenrud, 1995). There are to our knowledge no available surveys of amounts and types of building materials for Shanxi cities, and a specific study of material damage would necessitate substantial work with respect to a materials inventory.

With respect to vegetation damage, the level of SO_2 in many areas exceeds the guideline value of 20 and 30 μg/m^3 recommended by ECE to protect, respectively, natural vegetation and agricultural crops (UN-ECE, 1990). Moreover, it has been shown that regional haze, which is associated with regional-scale air pollution, may result in a marked reduction in the solar irradiance, which subsequently leads to a reduction in crop yields due to reduced photosynthesis. According to estimates by Chameides et al. (1999) crop losses up to as much as 30% may be the result of regional haze in some areas in China.

There are inherently large uncertainties in all steps in analyses of this kind. It is therefore of crucial importance that uncertainties are communicated in a proper way. The uncertainty ranges that are given in Table 9 only reflect uncertainties in the dose-response functions. The probabilities of outcomes in the lower and upper bounds may be calculated for instance by Monte Carlo simulation (stochastic simulations). Monte Carlo simulation takes account of the uncertainty distributions of every input parameter and variable in the calculation sequence (the distributions, however, in many cases need to be subjectively estimated). The advantages of the approach are that one may avoid disputes over best point estimates, the risk estimates are associated with a quantitative measure of uncertainty, and it allows for quantitative evaluation of a possible conservatism in point estimates (see for instance Burmaster and Stackelberg, 1991). Rabl and Spadaro (1999) argued that many of the distributions are lognormal so that geometric means are appropriate. They demonstrate that the uncertainties in an impact pathway analysis typically lead to the ratio between upper and lower bounds in the confidence interval of the damage cost estimate for many end-points being as large as one or two orders of magnitude. The estimate of the total damage cost, being a sum of many contributions, approaches a normal distribution with a smaller relative standard deviation than the most uncertain term. Clearly, it is very important to perform sensitivity analyses. The purpose of a sensitivity analysis is to evaluate the robustness of the outcome of calculations. If the outcome is very sensitive to parameters and variables which are based on limited or uncertain data, the confidence in the results will be low. If there, on the contrary, is a good basis for determining the parameters to which the output is more sensitive, there should be a high degree of confidence in the estimates. In the calculation made here, the output was rather sensitive i.a. to assumptions about how emission reductions influence the population exposure, and this becomes a central issue of further work. Also, further work is needed to estimate the economic impacts of abatement options in a broader perspective and to elucidate synergy effects, and lack of such, between alternative environmental policies.

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Annex 1 – Maps of Shanxi

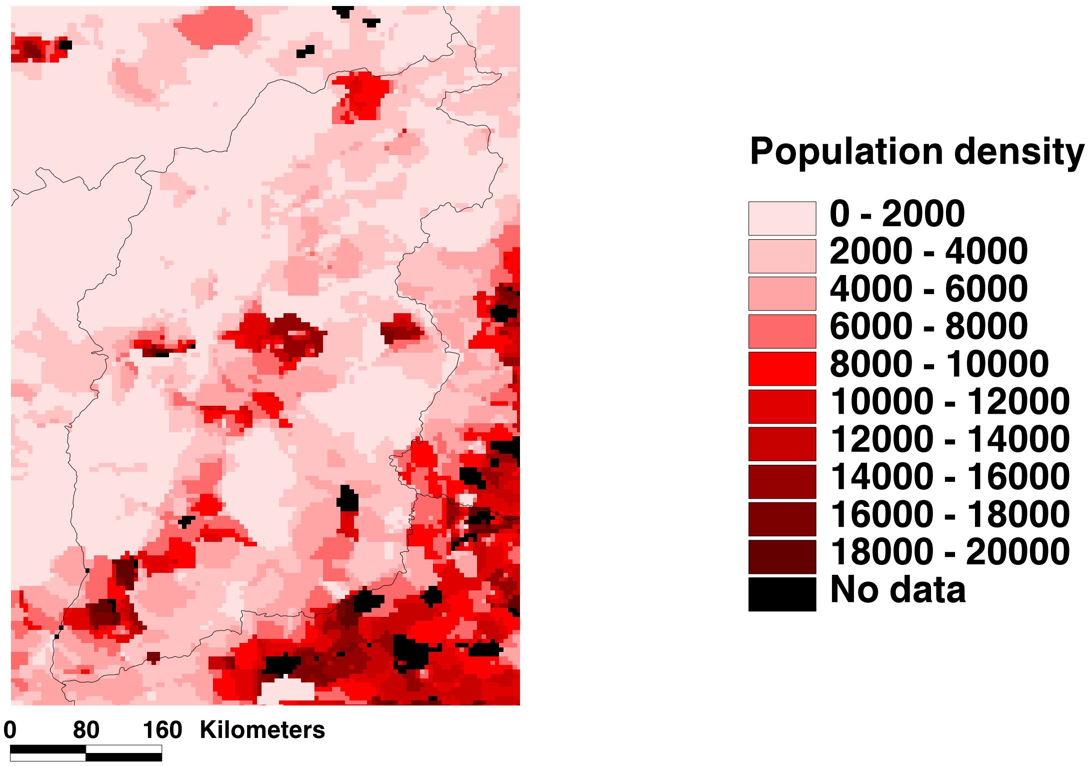

Figure A1. Population density in Shanxi (1990 figures). Data from UNEP (1996), see [*http://www.grida.no/prog/global/cgiar/asia/intro.htm*](http://www.grida.no/prog/global/cgiar/asia/intro.htm)

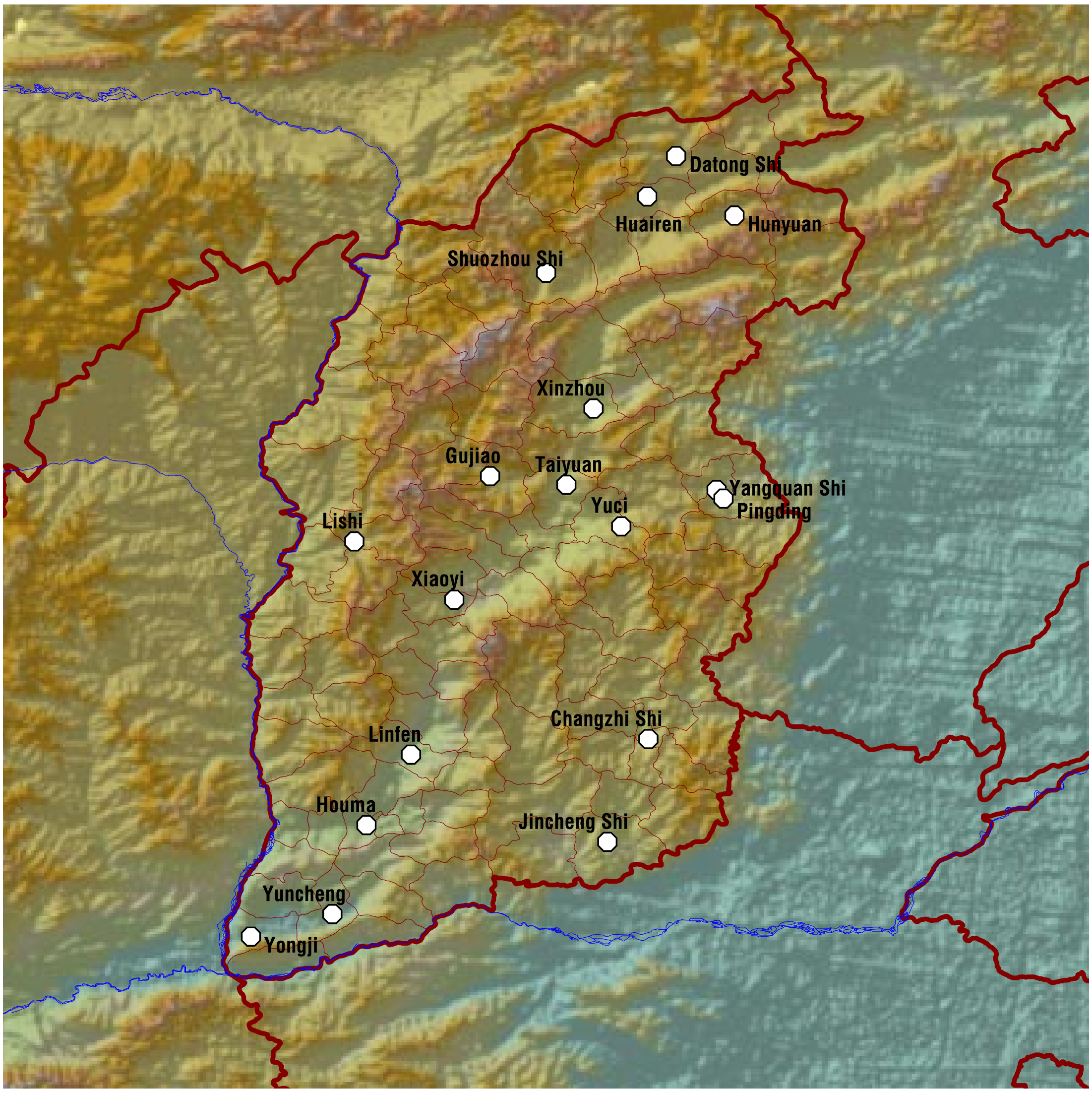

Figure A2. Topography of Shanxi, counties and location of 18 cities where air pollution data were available.

Annex 2 - Exposure-response functions applied in the Shanxi study

General

This annex gives an overview of some Chinese epidemiological papers that may be used to derive exposure-response functions. The reported relationships between air pollution and health effects are compared to corresponding literature from other parts of the world. For end-points not covered by Chinese studies, we rely on studies from Europe and the USA. The review is based on a work by Aunan and Li (1999) made in connection to a study i Guangzhou (see Aarhus, 1999). For some health end-points we did not have data for the present frequencies in Shanxi, hence a calibration of the functions for the conditions in Shanxi was not possible. In these cases we have, as a preliminary approach, used data from Guangzhou.

In the functions rendered in the following the estimated effect on health is attributed to one indicator component, i.e. we have used the reported results when regression models were fitted separately for the individual components. This does not imply that there is no effect of other air pollution components, but simply that each one of the components is treated as an indicator of the health damaging agent(s) in the pollution mixture. When calculating the health benefit from a certain reduction in air pollution one can therefore not add the estimates obtained by applying the PM_{10} and SO_2 functions, respectively, because each of them is supposed to indicate the total effect of the air pollution. In case a certain reduction is obtained both for PM_{10} and SO_2, we suggest to calculate the impacts by means of both sets of functions, and allow estimated damage to be the higher of the two estimates obtained. This is not a fully satisfactory procedure, but as long as independent functions in most cases are not available (and perhaps will never be due to the inherent synergistic effects between air pollutants) it is in our view justified. Another way of dealing with this methodological problem, as suggested by Wang and Kirk (1999), is to assume that a certain percentage of the SO_2 emissions are converted to fine particulates, and thereby estimate the health impact of sulfur reductions through their assumed contribution to reducing the level of particles.

We have applied a general PM_{10}/TSP conversion factor of 0.55 to estimate coefficients for PM_{10} when the original study reported TSP (US-EPA, 1982). There are some indications that this may be somewhat high for our purpose, but no data are to our knowledge available to suggest another factor. The uncertainty ranges given in the tables are tentative. Generally they indicate ±1SD. The present frequencies (i.e. number of cases per capita) of the health end-point given in the tables, as applied in the calibrated functions given in Table 6 above, do for all end-points but mortality (including infant mortality), represent the average for the eight central districts in Guangzhou (1996 data).

Causal agents in the air pollution mixture

From clinical studies it is known that several of the common air pollutants, as particles, SO_2, NO_2, and O_3, acid aerosols, and CO, have an adverse effect on human health when the concentration level exceed certain levels (see e.g. SFT, 1992). Generally, a large share of the epidemiological studies from the USA report that particles, or some exposure index related to particulate air pollution, have the best performance as the independent variable in exposure-response functions for several health end-points, although other air pollutants may also be associated with the effects. The evidence of particles playing a central role is especially strong in studies of mortality, for which quantitatively similar estimates have been reported over a wide range of concentrations, in a variety of communities with varying mixtures of pollutants and different climate. In European studies there is a tendency that more studies report associations also for other air pollutants than particles, especially for SO_2 (see e.g. Sunyer et al., 1996 and Vigotti et al., 1996) and for NO_2 (see e.g. Pönkä, 1991). In China, several studies report a clear effect of SO_2 on various health end-points (see below).

The effects of each of the air pollution components on human health may, however, be very difficult to disentangle. Especially particles, SO_2 and NO_2 may in many cases be regarded as an air pollution complex, since they are both chemically and biologically interrelated. The gases participate in the formation of aerosols and particles, the components are to a large extent associated with the same type of effects and synergism may occur. For instance, the dose of acidic components that enters the lungs may increase when fine particles are abundant in the air. The fine fraction (<2.5 µm) generally consists of soot and acid-condensates, mainly ammonium sulphates and ammonium nitrates. In addition, various organic chemicals, some of them potentially mutagenic, may be adsorbed onto carbon surfaces.

Whether it is the constituents or the size of the particles that is the more decisive for the health impacts, remains unknown, and may also differ for the various health effects. Whereas the coarser fractions, up to as large as 100 µm, indeed are important for some upper airway effects, the finer fractions are assumed to be better indicators for airway effects in general and effects in the smaller airways and alveoli in particular. Whereas PM_{10} now is commonly monitored in many countries and generally is held to be a reasonably good indicator in the context of health damage from air pollution, there has been a growing attention the recent years towards the impacts of the finer fractions of PM_{10}, e.g. $PM_{2.5}$ and the ultrafine particles (less than 0.1 µm in diameter). On a mass basis ultrafine particles constitute only a small share of PM_{10}, but they constitute a very large share of the surface area of an air sample due to their extremely large number. For equal masses of 10 µm and 0.01 µm particles, for instance, there will be 10^9 more particles of the smaller size (Williams, 1999). If the particles in the ultrafine range have a reactive surface, the large area for reaction with the lining fluids and cells of the respiratory tract provides a large potential for adverse effects. So far, the role of these particles in the adverse health effects of PM_{10}, however, remains unresolved (see Donaldson, Stone, and MacNee, 1999; Richard et al., 1999; and Ayres, 1998).

There is consistent evidence for health impacts of O_3, mainly in the lower respiratory system, independent from simultaneous exposure to particles or other components. Public health effect concerns for O_3 range from acute symptoms, decreased lung function and impairment of immune defence, to permanent scarring of lung tissue (fibrotic changes), which may be a factor in the development of chronic lung disease. Numerous studies, both in the USA and in Europe, report an association between health effects and ozone concentration (see US-EPA, 1996).

Chinese studies

Crude mortality rate

Whereas studies of the association between daily crude mortality rates and air pollution in the USA have indicated an approximately 0.1% increase in the rate per μg/m^3 PM_{10} (EC, 1995) studies in heavily polluted areas in developing countries generally report lower coefficients. For instance a study in Delhi (Cropper et al., 1997) reported an approximate increase of 0.04% per μg/m^3 PM_{10}. To our knowledge two studies on daily mortality rate have been done in Chinese cities, in Beijing (Xu et al., 1994) and in Shenyang (mentioned by Wells et al., 1994). According to Wells et al. (1994) the study in Shenyang reports a 0.02% (0.01%-0.04%) increase per μg/m^3 SO_2. The Beijing study reports a positive and significant log-linear relationship between crude mortality rate and SO_2, whereas the corresponding association for TSP is positive, but not significant. The study, however, report a significant association also for TSP during summer months. The authors suggest that air monitoring data, used to estimate population exposure, may reflect actual exposure more accurately in summer than in winter, because in winter indoor pollution is higher and the difference between indoor and outdoor concentration is likely to be higher. A possible misclassification of exposure will usually, but not necessarily, result in a downward bias of the observed association (Phillips and Smith, 1992).

Since we are not aware whether the Shenyang study has been published, we decided to omit the study as a basis for exposure-response functions, and suggest to linearise the log-linear functions for SO_2 and TSP found in Beijing over a quite wide concentration range. In Table A16 the SO_2 function is linearised over the range 50-300 μg/m^3, and the PM_{10} function is linearised over a TSP range of 200-400 μg/m^3. The estimated function for particles is in agreement with the above mentioned study in Delhi, where the average TSP in the study period was reported to be 378 μg/m^3, i.e. close to the level in Shanxi and Beijing. The estimate is also in accordance with studies in Europe, which indicate an increase in acute mortality of 0.04% per μg/m^3 PM_{10}. The function for SO_2, however, is somewhat higher than the relationship found in Europe, which indicates 0.07% increase per μg/m^3 SO_2 (see EC (1998) and references therein).

Table A16. Exposure-response functions for daily PM_{10} and SO_2 and daily crude mortality rate. Percentage increase in mortality rate per μg/m^3.

End-point	Indicator component	Coefficient	Annual crude mortality rate in Shanxi	Ref. (function).
Daily mortality rate	PM_{10}	0.036 (0-0.068)	0.006	Xu et al. 1994
	SO_2	0.12 (0.09-0.16)	“	Xu et al. 1994

Hospital outpatient visits and emergency room visits

Three studies of the association between air pollution and the daily number of various types of unscheduled hospital outpatients visits (i.e. polyclinical visits where the patient does not stay at the hospital during the night) have been performed in Beijing (Xu et al., 1995a, 1995b, and Dong et al., 1996). Functions for the total number of outpatient visits (OPV) and emergency room visits (ERV) derived by a meta-analysis of these studies are given in Table A17. The association between air pollution and OPV was in the studies showed to be stronger for some specific types of OPV (e.g. non-surgery and especially pediatric OPV). To simplify the benefit calculation, we chose to use the functions given for the total number of OPV. The functions reported by Xu et al. (1995b) and Dong (1995) are non-linear, and were in our calculation linearised over the range 200-400 $\mu g/m^3$ TSP. The study by Dong et al. showed a significant exposure-response function for TSP and non-surgery OPV, whereas SO_2 was significantly associated only with pediatric OPV. Concerning the particle functions for OPV and ERV the estimated coefficients are considerably lower than presumably comparable estimates from western studies (see e.g. EC, 1995). The SO_2 coefficient in the function for ERV is slightly lower than what has been estimated for Europe (Clench-Aas and Krzyzanovski, 1996).

The annual frequencies of hospital OPV and ERV per cap in Guangzhou are taken from the Statistical Yearbook of GZ (1998). Generally, ERV is more resource demanding than OPV, hence more costly in economic terms.

Table A17. Exposure-response functions for PM_{10} and SO_2 and the number of outpatient visits (OPV) and emergency room visits (ERV). Percentage increase in number of cases per $\mu g/m^3$.

End-point	Indicator component	Coefficient	Annual frequency in GZ[1]	Ref. (function).
Tot. OPV	PM_{10}	0.054 (0.023-0.086)	8.601	Xu et al. 1995a Xu et al., 1995b Dong et al., 1996
	SO_2	0.021 (0.0018-0.024)	“	Xu et al. 1995a Xu et al., 1995b Dong et al., 1996
ERV	PM_{10}	0.011 (0.003-0.020)[2]	0.499	Xu et al. 1995a
	SO_2	0.037(0.023-0.052)	“	Xu et al. 1995a

[1]Source: Statistical Yearbook of Guangzhou, 1997. The figures apply to the 8 central districts in Guangzhou.
[2]This function was not significant.

Hospital admissions

Associations between air pollution and hospital admissions (HA) have been reported in numerous studies, mainly in North America (EC, 1995). Typically, the association is reported to be closer for HA due to specific types of diseases that are connected to air pollution than for the total number of HA, for instance respiratory HA and HA due to chronic obstructive pulmonary diseases.

A study of the association between HA and air pollution has been done i Beijing (see Wells et al., 1994 for a summary of the results in English; the original study is published in Chinese). The exposure-response function from this study is given in Table A18. In Guangzhou the *average length of stay* for HA due to all kinds of diseases that are associated with air pollution is *21 days* (unpublished data from a study at four hospitals in Guangzhou).

Table A18. Exposure-response functions for PM_{10} and SO_2 and hospital admissions (HA). Percentage increase in number of cases per μg/m³.

End-point	Component	Coefficient	Annual frequency in GZ[1]	Ref. (function).
Tot. HA	PM_{10}	0.11 (0.07-0.14)	0.089	Xu et al., 1995b
	SO_2	0.21 (0.10-0.34)	“	Xu et al., 1995b

[1]Source: Statistical Yearbook of Guangzhou, 1997.

Chronic respiratory symptoms

A study of the effect of indoor and outdoor particulate level on chronic respiratory symptoms (CRS) in adults was performed by Xu and Wang (1993) in Beijing. The symptoms were chronic cough and phlegm, bouts of cough and phlegm, shortness of breath and wheeze. An analysis of the independent effect of particulate air pollution and SO_2 indicated that the association with particulates was much stronger than with SO_2, but that some of the effect may be attributable to SO_2.

In deriving the exposure-response estimate for the effect of outdoor particulate pollution we applied the reported 5 year mean exposure. The obtained function gives an indication of the probability of developing CRS during a 5 years period. To be able to estimate the average annual change in probability of developing CRS (during an approximate 5 years period) we divided the estimated regression coefficient by 5, see Table A19. The estimated percentage increase per year is quite close to what was found in a study by Abbey at al., (1993) in the USA (see also Ostro, 1996). There, however, the effect of particles on the risk of developing CRS (bronchitis specifically or various kind of airway obstructive diseases in general) was estimated over a 10 years period. Moreover, a study by Portney and Mullahy (1990) showed a somewhat higher annual percentage increase over a six years period. It is difficult to know from these studies what is the actual lag between a change in exposure and the full effect in terms of reduced risk in the population, and we decided to use the 5 year period since this was originally reported in the study in Beijing.

The Beijing study did not provide estimates of the response in children. In the light of a large study from USA which shows an effect of particulate air pollution on CRS in children (Dockery et al., 1993), we may assume that the function given in Table A19 can be applied to the total population. Unpublished data from an interview study in Guangzhou indicate a prevalence rate of CRS in children of about 7% (6% for bronchitis and 8-10% for lower respiratory symptoms) and about 5% in adults (2% for bronchitis and 8% for lower respiratory symptoms). These rates are used to estimate the calibrated functions in Table 6 in the main report, which gives the additional *new* cases per mill. per μg/m³ PM_{10}.

Table A19. Exposure-response function for PM_{10} versus annual new cases of chronic respiratory symptoms (CRS). Percentage increase in cases per μg/m³.

End-point	Indicator component	Coefficient	Present annual prevalence rate in GZ	Ref. (function).
CRS	PM_{10}	0.09 (0.08-0.10)	0.05-0.06	Xu and Wang, 1993

Endpoints not covered by Chinese studies

Infant mortality rate

In areas with relatively high air pollution levels an association between the infant mortality rate and air pollution have been reported (Bobak and Leon, 1992, Woodruff et al., 1997 and Joyce et al., 1989). A study at Taiwan reported a strong relationship between the rate of sudden death infant syndrome and visibility, where visibility was highly correlated with PM_{10} concentration (Knöbel et al., 1995).

The strongest association between infant mortality and air pollution is reported for mortality due to respiratory causes. The proposed function for PM_{10} given in Table A20 is based on a meta-analysis of the study by Bobak and Leon from the Czech Republic and the study done in the USA by Woodruff et al., and applies to crude infant mortality (not disease specific). In the first study no effect was found for SO_2 when the effect of PM_{10} and NO_x was adjusted for. In the meta-analysis we use the results for PM_{10} for which the effects of other pollutants were not adjusted for, hence PM_{10} should serve as an indicator of the air pollution mixture. The air pollution levels were lower in both studies than the present level in Shanxi. The study by Bobak and Leon (1992) was used to derive a function for SO_2. No effect on postneonatal mortality was shown below 45 $\mu g/m^3$, which could be interpreted as a threshold level.

The present birth rate in Shanxi is about 16.1 per 1000 inhabitants. The infant mortality rate is 13.7 per 1000 live births (National Bureau of Statistics, 1999). These figures are used to derive the calibrated functions in Table 6 in the main report.

Table A20. Exposure-response functions for PM_{10} and the infant mortality rate (postneonatal mortality, deaths per live births). Percentage increase in cases per $\mu g/m^3$.

End-point	Indicator component	Coefficient	Present rate in Guangzhou[1]	Ref. (function).
Infant mortality	PM_{10}	0.55 (0.35-0.76)	0.016	Bobak and Leon, 1992 Woodruff et al., 1997
	SO_2	0.15 (-0.50-0.82)	"	Bobak and Leon, 1992

[1]Deaths per live births

Restricted activity days and work day losses

Relatively few studies have been performed regarding how air pollution influences sick-leave frequencies and other kinds of reduced labour productivity. A study in Oslo, where the level of air pollutants is rather low, indicated a percentage increase in the number of sick-leaves of 0.6% per $\mu g/m^3$ PM_{10} (Hansen and Selte, 1997).

Ostro (1987) analysed the relationship between sick-leaves, called work day losses (WDL), and so-called restricted activity days (RAD) in 49 metropolitan areas in the USA during six years. Concerning WDL the study indicates a 0.4% increase per $\mu g/m^3$ PM_{10}. The result for RAD is regarded as being more certain, and is often being used in applied studies. The increase in RAD per $\mu g/m^3$ is estimated to be 0.3% (see Ostro, 1996).

In the case of Guangzhou and Shanxi we have no information of the frequency of RAD. From the interview study, however, we know that the average annual frequency of sick-leave days per person is 6.3. We suggest using this and the function for WDL from Ostro (1987).

Outpatient visits, emergency room visits, and hospital admissions entail work day losses or absence from ordinary tasks and duties. In calculating the economic benefit related to these end-points the cost related to the work day loss should therefore be excluded, if the function for WDL is applied in addition.

Table A21. Exposure-response function for PM_{10} and work day losses (WDL). Percentage increase in cases per $\mu g/m^3$.

End-point	Indicator component	Coefficient	Present rate in Guangzhou[1]	Ref. (function).
WDL	PM_{10}	0.4 (0.2-0.6)	6.3	Ostro, 1996

[1] Annual number of sick-leave days per adult (unpublished data from an interview study).

Acute respiratory symptoms

Respiratory symptoms and diseases may be subdivided in various ways. One may distinguish between upper and lower respiratory symptoms (URS and LRS). Usually one may assume that a case of ARS last for one day – called one "symptom day". The distinction between URS and LRS is not quite clear, e.g. for cough, but generally symptoms as coryza, runny/stuffed nose, sneezing and sore throat are defined as URS, whereas e.g. wheeze, chest tightness, shortness of breath, and phlegm production are classified as LRS. URS are generally far more frequent, but less severe in terms of entailing activity restriction.

In the following proposed functions for acute respiratory symptoms in general (ARS) no distinction between URS and LRS is made. The function for ARS in children and particles is based on a meta-analysis of three studies from Europe and the USA (see Aunan, 1996). A corresponding function for SO_2 may be based on a USA study by Schwartz et al. (1991), which had cough in children as the end-point (see also Ostro, 1994). The functions for ARS in adults are based on two USA studies by Krupnick et al., (1990) and Schwartz et al. (1988), the latter reporting the effect of SO_2 on "chest discomfort", most likely an indicator of lower respiratory symptoms. Generally, the air pollution level in the studies were well below the concentrations seen in Guangzhou.

Whereas the estimated effects in children and adults for particulate air pollution seem reasonably coherent, the effect of SO_2 is very different for the two age groups. One reason may simply be that the types of respiratory symptom analysed are too different to represent a general case of ARS. In light of other studies showing an effect on ARS in children of SO_2 (e.g. Dodge et al., 1985) it seems likely that the function for SO_2 and ARS in children given in Schwartz et al. (1991) and Ostro (1994) understates the response. We suggest to apply the function for ARS in adults and SO_2 also for children. This is done in Table 6, where different prevalence rates of ARS in two age groups were assumed in the calibration. Based on data from the interview study mentioned above we may assume a prevalence rate of about 4% in both children adults (refers to the prevalence rates for sore throat, coughing and other respiratory symptoms).

Table A22. Exposure-response function for PM_{10} and acute respiratory symptoms (ARS). Percentage increase in cases per μg/m³.

End-point	Indicator component	Coefficient	Present annual frequency in GZ	Ref. (function).
ARS in children	PM_{10}	0.55 (0.36-0.82)	0.04[1]	Aunan, 1996
ARS in adults	PM_{10}	0.27 (0.20-0.33)	0.04[1]	Krupnick et al., 1990[2]
	SO_2	0.072 (0.068-0.075)		Schwartz et al. 1988[2]

[1] In the studies in USA this was about 0.07 for children and about 0.03 for adults.
[2] See also Ostro (1994).

Asthma attacks

Asthmatics are generally held to be more sensitive to air pollution than non-asthmatics, and several studies have showed a relationship between elevated asthma symptoms and particulate air pollution (e.g. Whittemore and Korn, 1980, Ostro et al., 1991, and Forsberg et al., 1993). Asthma attacks are also reported to be associated with the level of SO_2 and NO_2 (e.g. Bates et al., 1990 and Pönkä, 1991).

To calculate the change in number of asthma attacks per asthmatic person we rely on Ostro (1996), which indicates that the annual number of asthma attacks per asthmatic increase by 0.06 (0.03-0.20) per μg/m³ PM_{10} (annual mean). The share of the population in Guangzhou being asthmatics, as estimated from the interview study, is about 3%. This is used in the function given in Table 6 in the main report.

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Visiting address: Sognsveien 68, Oslo
Telephone: +47 22 85 87 50 Fax: +47 22 85 87 51
E-mail: admin@cicero.uio.no Web: www.cicero.uio.no

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CICERO Senter for klimaforskning
Postboks 1129 Blindern, 0318 Oslo
Besøksadresse: Sognsveien 68, Oslo
Telefon: 22 85 87 50 Faks: 22 85 87 51
E-post: admin@cicero.uio.no Hjemmeside: www.cicero.uio.no