The Challenge of Air Pollution

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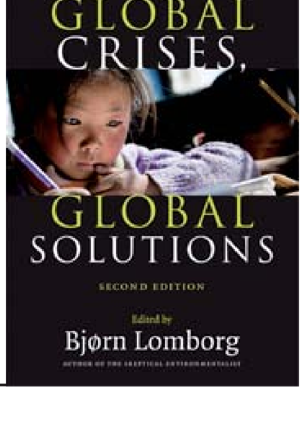

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Air Pollution

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INTRODUCTION

Air pollution in its broadest sense refers to suspended particulate matter (dust, fumes, mist and smoke), gaseous pollutants and odors (Kjellstrom et al., 2006). To this may be added heavy metals, chemicals and hazardous substances. A large proportion of air pollution worldwide is due to human activity, from combustion of fuels for transportation and industry, electric power generation, resource extraction and processing industries, and domestic cooking and heating, among others. Air pollution has many impacts, most importantly affecting human and animal health, buildings and materials, crops, and visibility.

In addressing the multiple burdens of air pollution, its related causes, and the solutions, a broad distinction is necessary between indoor and outdoor air pollution:

- Human-induced ***indoor air pollution*** is to a large extent caused by household solid fuel use (SFU) for cooking and heating, usually involving open fires or traditional stoves in conditions of low combustion efficiency and poor ventilation. Indoor air pollution also originates from other "modern" indoor air pollutants associated with industrialization, with a variety of suspected health effects such as sick-building syndrome. However, from a global burden of disease point of view, these modern indoor air pollutants are relatively minor; hence this study focuses on air pollution from SFU. Due to the close proximity and low or zero cost of solid fuels such as biomass in most rural areas, indoor air pollution is more of an issue in rural than in urban areas, although in many urban areas coal and charcoal are common household energy sources. Indoor air pollution from SFU is particularly hazardous given that pollution concentrations often exceed WHO guidelines by a factor of 10-50. Indoor air pollution is also related to environmental tobacco smoke ('passive smoking') and exposure to chemicals and gases in indoor workplaces.

- Human-induced ***outdoor air pollution*** occurs mainly in or around cities and in industrial areas, and is caused by the combustion of petroleum products or coal by motor vehicles, industry, and power generation, and by industrial processes. Outdoor air pollution is fundamentally a problem of economic development, but also implies a corresponding underdevelopment in terms of affording technological solutions that reduce pollution, availability of more energy-efficient public transport schemes, and enforcing regulations governing energy use and industrial emissions.

Rates of exposure to these two types of air pollution therefore vary greatly between rural and urban areas, and between developing regions, given variations in vehicles ownership and use, extent and location of industrial areas and power generation facilities, fuel availability, purchasing power, climate and topology, among others. Indoor sources also contribute to outdoor air pollution, particularly in developing countries; vice versa outdoor air pollution may contribute to pollution exposure in the indoor environment (Kjellstrom et al., 2006).

Over 3 billion people are exposed to household air pollution from solid fuels used for cooking and heating, and over 2 billion people are globally exposed to urban air pollution in more than 3,000 cities with a population over 100 thousand inhabitants.[1] Epidemiologically, household SFU and urban air pollution differ in important respects. SFU is disproportionately affecting young children and adult females, while urban air pollution, according to current evidence and assessment methods, is predominantly affecting adults and especially the older population groups. There are also important differences in terms of solutions. Air pollution from SFU can be substantially reduced or practically eliminated by a few interventions such as installation of improved stoves with chimney or a substitution to "clean" fuels such as liquefied petroleum gas (LPG), natural gas, or, potentially, biomass gasifier stoves. However, broad packages of interventions are often required to achieve any significant improvement in urban air quality.[2] Given these differences, this paper discusses SFU and urban air pollution separately.

While there are many air pollutants, current assessment methods identify fine particulates (PM2.5) as the pollutant with the largest health effects globally. The focus of this paper is therefore particulates. Particulates are caused directly by combustion of fossil fuels and biomass, industrial processes, forest fires, burning of agricultural residues and waste, construction activities, and dust from roads, but also arise naturally from marine and land based sources (e.g. dust from deserts). Particulates, or so called secondary particulates, are also formed from gaseous emissions such as nitrogen oxides and sulfur dioxide.

[1] The World Bank provides air quality modeling results for these cities. They are therefore used here as an indicator of global population exposed to urban air pollution.
[2] An exception is elimination of lead (Pb) from gasoline, or control of localized pollution from industrial plant(s) or thermal power plant(s).

HOUSEHOLD AIR POLLUTION FROM SOLID FUELS

1. The Challenge

An estimated 1.5 million deaths occur annually as a result of household air pollution from SFU mainly for cooking as well as winter season heating. The total disease burden, including morbidity, is estimated at 36 million DALYs (WHO 2007).[3] These deaths and DALYs arise mainly from acute lower respiratory infections (ALRI) in young children and chronic obstructive pulmonary disease (COPD) in adults, and to a lesser extent lung cancer. There is also moderate evidence of increased risk of asthma, cataracts and tuberculosis (Desai et al, 2004; Smith et al, 2004). While urban air pollution is strongly associated with elevated risk of heart disease and mortality (Pope et al, 2002), no credible studies of such a link are available for SFU because of the longitudinal data requirements. It is however plausible that SFU is a contributor to heart disease and mortality, and, if so, health effects of SFU might currently be significantly underestimated.

By WHO region of the world, use of improved domestic fuels (e.g. LPG, kerosene) in rural areas vary from under 15 percent in Sub-Saharan Africa and South East Asia, to 33 percent in the Western Pacific developing region, and closer to 50 percent in Eastern Mediterranean and Latin American countries. The main types of unimproved fuels used in rural areas are firewood, dung and other agricultural residues, followed by charcoal and coal/lignite (Rehfuess et al., 2006). Indoor air pollution from SFU is generalized throughout the developing world. However, the health effects depend on many factors, including type of solid fuel and stove, household member exposure to solid fuel smoke (e.g. household member activity patterns, indoor versus outdoor burning of fuels, cooking practices and proximity to stove, and smoke venting factors such as dwelling room size and height, windows and doors, construction material, chimney), and household member age and baseline health status and treatment of illness.

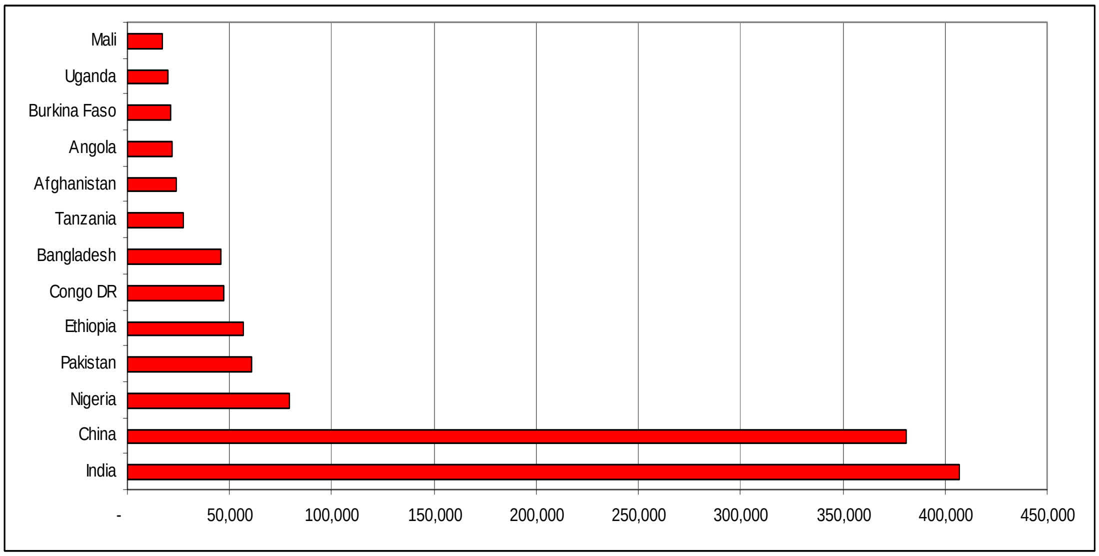
About 1.2 million or 80 percent of global deaths from SFU occur in 13 countries. Eight of these countries are in Sub-Saharan Africa and five are in Asia. India and China alone account for over 50 percent of global deaths from SFU (figure 1.1). Average prevalence of household SFU is over 90 percent in these 13 countries, ranging from 67 percent in Nigeria, 70 percent in Pakistan, about 80-82 percent in China and India, 89 percent in Bangladesh and over 95 percent in eight of the other countries. With the exception of China, these countries are characterized by relatively high u5 child mortality rates, high malnutrition rates, and low national income levels (table 1.1).

Larsen (2007a) provides an estimate of mortality from indoor air pollution from household solid fuels in rural China. The central estimate of annual mortality is 460 thousand assuming 50 percent of solid fuel stoves have a chimney and 355 thousand if 100 percent of solid fuel stoves have a chimney, suggesting that mortality from SFU in

[3] Estimated using baseline health data for the year 2002 and most recent available data on prevalence of household SFU.

China may be somewhat higher than presented in figure 1.1. The estimates are based on the same health end-points as in Smith et al. (2004) and WHO (2007). A framework with multi-level risks is applied to reflect some of the diversity of solid fuels and stove and venting technologies commonly used in households in China. Seven indoor air pollution exposure and risk levels are applied: households using predominantly biomass with or without chimney, a combination of biomass and coal with or without chimney, predominantly coal with or without chimney, and households using non-solid fuels (mainly LPG).

Figure 1.1 Annual deaths from household SFU air pollution (year 2002)



Source: Produced by the author from national estimates by WHO (2007). Mortality estimates are adjusted by the author for Pakistan to reflect most recent data on prevalence of SFU.

Table 1.1 Profile of 13 countries with the highest mortality from SFU

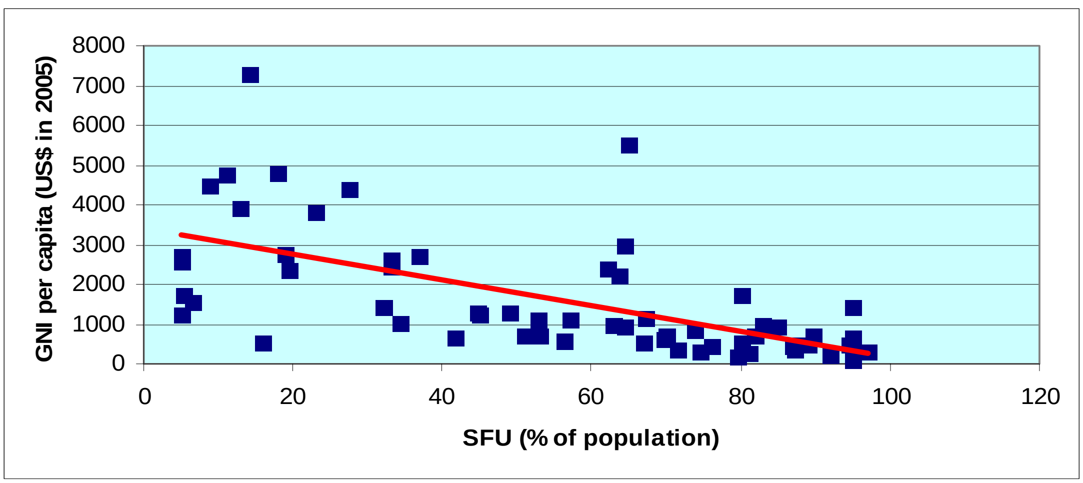
	India	China	Other countries (11 with highest mortality from SFU)
Average SFU prevalence (most recent available)	82%	80%	> 90%
Deaths from SFU in 2002	407,100	380,700	421,600
ALRI (% of deaths from SFU)	62%	5%	86%
COPD (% of deaths from SFU)	38%	90%	14%
LC (% of deaths from SFU)	0.1%	5%	0.01%
U5 child mortality rate in 2005	74	27	148
U5 child malnutrition (moderate and severe underweight)*	47%	8%	33%
GNI per capita in 2005	730	1,740	480

* Most recent data available from Unicef Global Database on Undernutrition.

An important question is if countries will grow themselves out of the SFU and associated health effects in the next few decades without a need for large scale interventions. One argument is that prevalence of household SFU is strongly correlated with country income level, so economic growth will solve the problem (figure 1.2). A second argument is that child mortality rates are declining so u5 mortality from SFU will gradually decline (by reducing ALRI case fatality rates) even without a reduction in SFU. A counter-argument is however that COPD mortality could possibly increase with aging populations even with a gradual decline in SFU. Each of these issues deserves attention and a set of simple projections are therefore presented in this paper.

A linear regression analysis shows that an increase of US $1,000 in GNI per capita is associated with a 20 percentage point decline in SFU prevalence. Let us assume that this cross-country relationship holds intertemporally for the 13 countries that account for 80 percent of SFU mortality. In the 11 countries other than China and India in figure 1.1, it would take about 55 years to reduce SFU prevalence to 50-55 percent and 75 years to reduce SFU prevalence to 10 percent, at a per capita income growth of 3 percent per year. In China and India it would take 10-20 years and 20-30 years, respectively, at current economic growth rates. However, SFU prevalence in China has not declined at a rate anywhere close to the rate suggested by the cross-country regression results, although a substantial substitution from fuel wood to coal has been observed in the last couple of decades. Fuel substitution has also been quite slow in India despite rapid economic growth in the last decade.

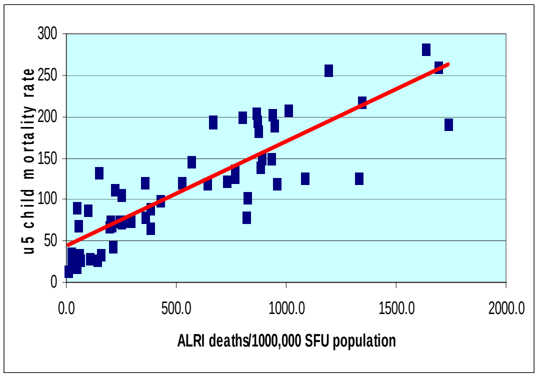
Figure 1.2 Household SFU prevalence rates and GNI per capita

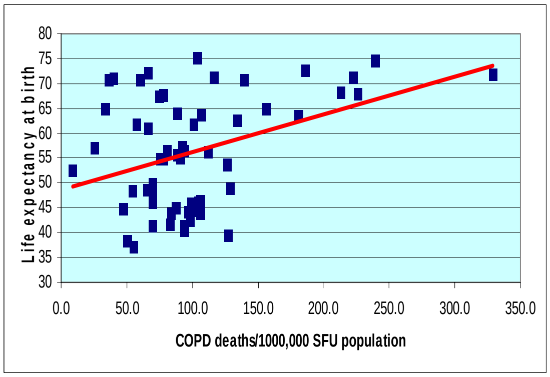


Source: The author. GNI per capita is from WDI 2007. SFU is from WHO (2007).

In most countries, a majority of deaths from SFU is mortality from ALRI in children u5. There is a strong correlation between SFU deaths per population and u5 child mortality rates. COPD mortality is to some extent correlated with life expectancy and an aging population (figure 1.3).

Figure 1.3 Deaths from SFU in relation to child mortality rates and life expectancy





Source: Prepared by the author. U5 child mortality rate and life expectancy at birth are for 2005 (World Bank, 2007). ALRI and COPD deaths from SFU are from WHO (2007). Countries with >= 1000 deaths from SFU are included in the chart.

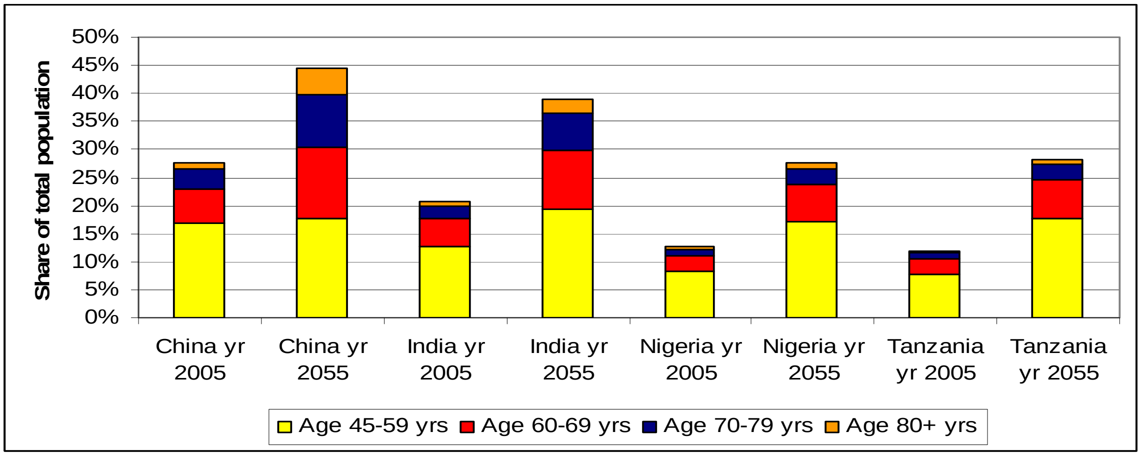
ALRI mortality from SFU has most likely declined in the last decades, and is likely to decline further even without a reduction in SFU or adoption of improved stoves. This comes about from a reduction in ALRI case fatality rates through for instance improved case management and reduction in malnutrition rates even in the event that incidence of morbidity does not decline.[4] In the countries with the highest SFU mortality (in the sample of 13 countries), u5 child mortality rates have declined substantially since 1960 but appear to have stagnated in several of the Sub-Saharan countries. At rates of decline observed in the last 2 decades, it would take an average of 35 years in Bangladesh, India and Pakistan for u5 child mortality rates to reach the current rate of 27 per 1000 live births in China. It would take an average of 75 years in Ethiopia, Uganda and Tanzania.[5]

If all-cause ALRI mortality declines at the same rate as u5 child mortality, and there is no change in SFU, then in 50 years annual ALRI mortality from SFU would be 250 thousand, or 40 percent of the current level in this group of 13 countries.

COPD mortality occurs largely in older population groups. With aging of populations over time, COPD mortality from SFU could increase over the next 50 years. The share of population aged 45+ years is expected to nearly double in China and India and more than double in Nigeria and Tanzania from year 2005 to 2055. The fastest growth in China and India is expected to be for the population aged 60+ (figure 1.4).

[4] See Fishman et al. (2004) for a discussion of child mortality risk in relation to malnutrition.
[5] This calculation is based on average u5 mortality rates and rates of decline in the groups of countries. Years required to reach the level of China will be different in each individual country.

Figure 1.4 Demographic projections 2005-2055



Source: Prepared by the author using World Bank demographic projections.

To provide a simple projection of COPD mortality from SFU, consider a scenario in which age-specific COPD death rates (per 1000 population in age group) are constant over time.[6] Using World Bank country demographic projections, we can apply the relative risks of COPD from SFU in Desai et al (2004) to estimate COPD mortality by SFU prevalence rates in 50 years from now. The results are presented for China, India, Nigeria and Tanzania in tables 1.2.

COPD mortality from SFU would be higher in 2055 than today in all four countries at SFU prevalence rates > 25 percent in year 2055 (current SFU prevalence is 67 to 95+ percent). SFU needs to decline to < 15 percent in Nigeria for COPD mortality to fall below today's level (table 1.3). The main drivers of these projections are aging of the population and population growth. But even COPD death rates (COPD deaths/population) would be higher than today unless SFU prevalence falls below 25-30 percent in China and Nigeria and below 35-40 percent in India and Tanzania. Assuming that SFU cross-country income elasticities are realistic, income growth alone would not alleviate any or much of COPD mortality from SFU.

[6] Age-specific COPD death rates are taken from Global Burden of Disease regional tables.

Table 1.2 Projections of COPD deaths from SFU

SFU prevalence in 2055	COPD deaths: Ratio of deaths in yr 2055/yr 2005			
	China	India	Nigeria	Tanzania
0.6	2.67	2.52	4.10	2.77
0.5	2.23	2.10	3.42	2.31
0.4	1.78	1.68	2.74	1.85
0.3	1.34	1.26	2.05	1.39
0.2	0.89	0.84	1.37	0.92
0.1	0.45	0.42	0.68	0.46
0	0.00	0.00	0.00	0.00

Source: Author.

2. The Solutions

There exists a range of solutions to reduce exposure to indoor air pollution. This includes reducing the source of pollution and altering the living environment and user behavior. Source reduction involves improved cooking devices (with or without flue attached), cleaner fuel, and reduced need for fire. Alterations to the living environment include improved ventilation and improved kitchen design and stove placement. Altered user behavior includes fuel drying, stove and chimney maintenance, use of pot lids to conserve heat, and keeping children away from the smoke (Bruce et al., 2005).

While there are many options available for reducing exposure to indoor air pollution, there is limited evidence on their effectiveness in real-life conditions for modeling the cost-benefit of these options. These include behavioral dimensions such as location of cooking area (indoor vs. outdoor; separate indoor area) and location of young children in relation to cooking area (carrying babies while cooking; playing near cooking area). Benefits of these behavioral modification are however difficult to quantify and depend very much on particular circumstances. Hence the solutions to household SFU air pollution that lend themselves to a cost-benefit analysis fall into two categories: (a) improved stove technology; and (b) substitution to cleaner fuels. The focus of this paper is therefore on technology and fuel choice.

Some results of indoor particulate (PM) concentrations measurements in relation to type of stove and fuel from Latin-America are presented in table 2.1. The improved stoves, such as the *plancha*, produce PM 2.5 or PM 3.5 levels that are often only 20 percent of concentration levels from an open fire, and are even found to be less than 10 percent of that of an open fire in a study in Guatemala by McCracken and Smith (1998). The reduction in PM 2.5 seems to be even larger than reductions in PM 10. However the concentration levels of PM, even with an improved stove, are still substantially higher than found in most outdoor urban environments and many times higher than the WHO guidelines for ambient PM concentrations (table 2.2). It may also be noted that although the use of liquefied petroleum gas (LPG) eliminates PM from fuel sources, indoor PM may still be significant do to other sources of pollution.

Table 2.1 Indoor particulate (PM) concentrations from cooking stoves

	Open fire/ Traditional Stove	Improved Stove	LPG	
24-hour PM 3.5	1930	330	-	Guatemala. Albalak et al (2001).
24-hour PM 10	1210	520	140	Referenced in Albalak et al (2001), adapted from Naeher et al (2000).
24 hour PM 2.5	520	88	45	
24-hour PM 2.5	868	152	-	
PM 10	600-1000	300	50	Mexico. Saatkamp et al (2000)

Source: Reproduced from Larsen (2005).

Table 2.2 WHO air quality guidelines

	Annual average (ug/m3)	24-hour average (ug/m3)
PM2.5	10	25
PM10	20	50

Source: NAE/NRC (2008).

While wood and to some extent charcoal are the most common solid fuels used in developing countries, China and Mongolia have high household prevalence of coal, especially for heating in open portable space heaters, some with and some without chimney. Mestl et al. (2006), based on data from the China 2000 Population Census, report that about 60 percent of rural households used biomass as primary cooking fuel. Nearly 30 percent of rural households used coal as the primary fuel. Mestl et al model annual average population weighted exposure (PWE) to indoor air pollution by using monitoring data reported in Sinton et al (1995) and a few recent studies, the share of the population using solid fuels, and household member activity patterns.[7] PWE to PM_{10} indoor air pollution is estimated at 360 ug/m^3 for rural households using coal and 810 ug/m^3 for rural households using biomass (figure 2.1).[8] Most of the monitoring studies of indoor PM_{10} in Sinton et al used by Meastl et al are however from the late 1980s and early 1990s and may therefore not reflect current indoor PM levels.

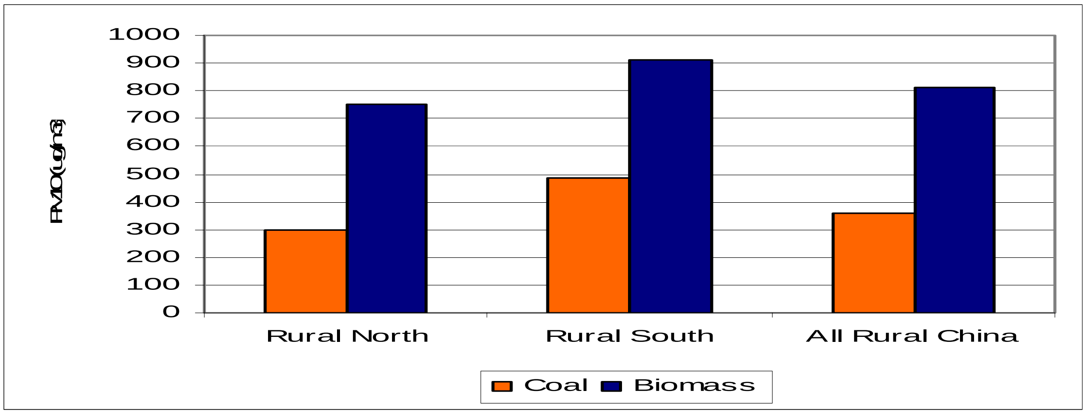
In a recent study by the World Bank, China CDC and other institutions in China, indoor air quality was monitored in select rural households in four of the poorest provinces in China. Three of these provinces are in northern China (Jin et al 2005). In this study, indoor PM_4 levels in households using predominantly biomass are roughly twice the level in households using predominantly coal (table 2.3).[9] This is very similar to Meastl et al (2006). Jin et al do however report somewhat lower concentration levels in the rural

[7] Activity patterns: Mestl et al estimate time spent per day for individuals in various age groups in several indoor microenvironments (kitchen, bedroom, living room, and indoors away from home) and outdoors.
[8] Actual indoor air PM_{10} is however higher than these levels because PWE adjusts for time spent outdoors where pollution levels are lower.

[9] Mestl et al reports PM_{10} concentration levels while Jin et al reports PM_4. These two measures of PM are not directly comparable as PM_4 is a subset of PM_{10}.

North than found by Meastl et al. But when compared to provincial/county level modeling results in Meastl et al the results are of similar orders of magnitude.

Figure 2.1 Population weighted exposure to indoor particulates (PM_{10})



Source: Larsen (2007a), produced from data presented in Mestl et al (2006).

Table 2.3 PM_4 Concentrations in Rural Households in China (ug/m^3)

Province	**Gansu**		**Inner Mongolia**	**Guizhou**		**Shaanxi**	
Main cooking fuel	Biomass		Biomass	Coal		Coal and biomass	
Main heating fuel	Biomass		Biomass and coal	Coal		Coal	
Time period*	Mar	Dec	Dec	Mar	Dec	Mar	Dec
Location/							
Cooking room	518	661				187	223
Living/bedroom	351	457					
Living room						215	329
Bedroom				315	202	186	361
Cooking/living room				352	301		
Cooking/living/bedroom			719				

Source: Reproduced from Jin et al (2005). * Time period of PM measurements was March and December.

Edwards et al (2006) report indoor air quality monitoring in nearly 400 households in the three provinces of Hubei, Shaanxi, and Zhejiang in 2002-2003. Great care was taken to select homes that reflect the diversity of fuels and stove technology and stove performance in China. PM_4 concentrations in 75 percent of kitchens and 73 percent of living rooms during the winter - and 48 percent of kitchens and 46 percent of living rooms during the summer - exceeded the national indoor air quality PM_{10} standard of 150 ug/m^3 for a 24 hour average. If PM_{10} had been measured, a greater the percentage of homes would have exceeded the standard in both seasons.

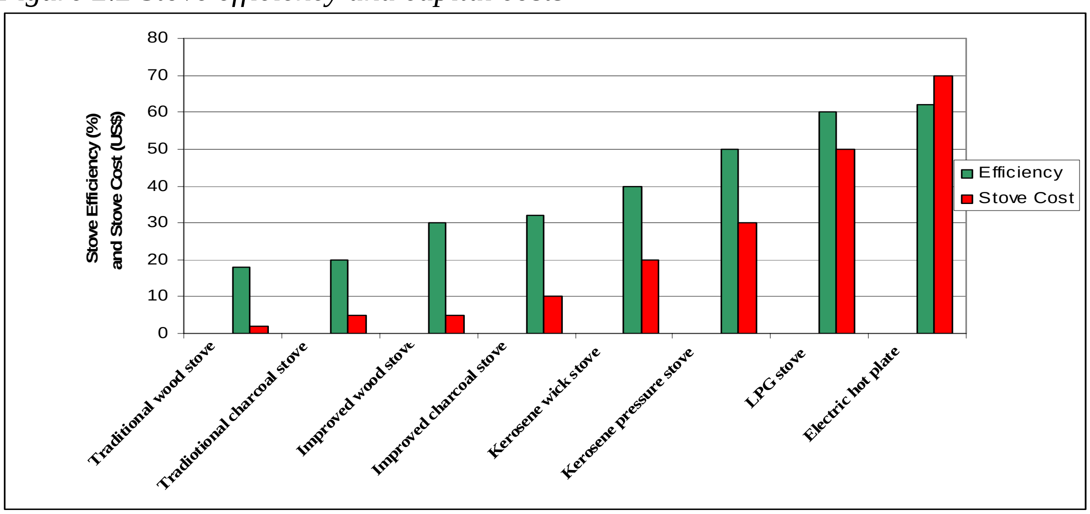
Edwards et al conclude that PM_4 concentrations are significantly lower in the homes with improved stoves (chimney) -- 152 ug/m^3 compared to 268 ug/m^3 in homes with unimproved stoves (no chimney). The study is however not conclusive regarding PM concentrations associated with coal versus biomass fuel, for similar types of stove technologies.

Ezzati and Kammen (2001, 2002) present indoor PM concentration measurements also for several types of charcoal stoves. PM concentrations are found to be substantially lower than concentrations from fuel wood stoves. Charcoal is often considered by households as an intermediate fuel on the energy ladder. While it certainly is not considered a clean fuel, it is often a preferred choice in many urban areas whenever available instead of fuel wood, before households can afford LPG or other clean alternatives.

In some Sub-Saharan countries the use of charcoal is relatively widespread especially in urban areas. Charcoal is used by 14-36 percent of households in declining order in Ghana, Congo Republic, Zambia, Madagascar, Tanzania, Uganda, Benin and Kenya.[10] In most South and East Asian countries, charcoal is much less prevalent with the exception of Thailand (19%; MICS, 2006) and increasingly in Cambodia (9%; DHS 2005). In Latin America, more than 40 percent of households in Haiti use charcoal, but the prevalence is very low in most of the other countries in the region.

Figure 2.2 presents an energy efficiency ladder for stoves, and their typical costs that is often sited in the research literature on fuel use and indoor air pollution (e.g. Baranzini and Goldemberg 1996; Luo and Hulscher 1999; and Saatkamp et al 2000). The stove efficiency ladder provides a generic perspective on potential energy savings from improved wood and charcoal stoves and kerosene, LPG, and electric stoves in comparison to traditional stoves. According to figure 2.2 improved wood and charcoal stoves are about 50 percent more efficient than traditional stoves, and LPG and electric stoves are 2 times more efficient than the improved wood and charcoal stoves.

Figure 2.2 Stove efficiency and capital costs



Source: Larsen (2005).

[10] From the most recently available Demographic and Health Surveys in Africa.

3. Economic Estimates of Costs and Benefits

Benefit-cost ratios of intervention to control or prevent air pollution from household SFU will depend on what benefits and costs are included in the analysis and how non-market benefits and costs are valued. Health effects of air pollution are often a major concern and motivation for intervention. Which health effects to include and how they are valued are therefore an important consideration in an economic analysis. Large scale household stove programs have also been motivated by natural resource considerations, for instance in China in the 1980s. The aim of that program was primarily improved energy efficiency but PM concentrations were also lowered (Edwards et al., 2006). Benefits may include environmental improvements and time savings from reduced fuel collection in addition to health benefits.

Desai et al. (2004) provide a meta-analysis of health effects from household solid fuel air pollution. Health effects were categorized by level of evidence from the research literature. Relative risk ratios associated with solid fuel use, relative to clean fuels such as LPG, were derived for each health outcome (table 3.1).[11] The national and global mortality and DALY estimates presented by WHO (2007) reflect the relative risk ratios in Desai et al, limiting the health effects to ALRI in children u5, and COPD and lung cancer in adult women and men.

ALRI in children from SFU is important for at least two reasons. ALRI is much more severe and involves more sick days than for instance acute upper respiratory infections (URI), and respiratory child mortality is almost exclusively from ALRI. However, in countries with lower child mortality rates, the cost of morbidity relative to mortality increases. This is for instance the situation in China and much of Latin America. Definition of morbidity health end-points and relevant age groups for inclusion in an economic analysis is therefore important especially in these countries. While ALRI is a major concern, URI and other respiratory symptoms are however far more frequent than ALRI.

Table 3.1 Relative risk ratios from a meta-analysis of research literature

Evidence	Health Outcome	Population group	Relative Risk	Confidence Interval
Strong	ALRI	Children < 5	2.3	1.9 – 2.7
	COPD	Women >= 30	3.2	2.3 – 4.8
	Lung cancer*	Women >= 30	1.9	1.1 – 3.5
Moderate – I	COPD	Men >= 30	1.8	1.0 – 3.2
	Lung cancer*	Men >= 30	1.5	1.0 – 2.5
Moderate - II	Lung cancer**	Women >= 30	1.5	1.0 - 2.1
	Asthma	Children 5-14	1.6	1.0 – 2.5
	Asthma	All >= 15	1.2	1.0 – 1.5
	Cataracts	All >= 15	1.3	1.0 – 1.7
	Tuberculosis	All >= 15	1.5	1.0 – 2.4

Source: Desai et al. (2004). * exposure to coal smoke; ** exposure to biomass smoke.

[11] The relative risks largely reflect the use of unimproved wood and coal stoves without chimney.

Several studies in China document the increased risk of respiratory illness and symptoms from SFU (table 3.2). Ezzati and Kammen (2001) find in Kenya that SFU air pollution substantially increases the risk of acute respiratory infections in general and not only ALRI. This is the case for both children and adult females, although the sample size in their study was relatively small (table 3.3).[12] Therefore quantifying the cost of these health end-points seem to be important.

Table 3.2 Relative risk ratios from recent studies of indoor air pollution in China

Health outcome or endpoint	Pollutant or fuel	Relative risk ratios*	Confidence interval (95%)	Notes	Reference
Bronchitis (acute)	Coal smoke (heating)	1.73 [a]	1.42 – 2.12	Children in urban Chongqing, Guangzhou, Lanzhou, and Wuhan.	Qian et al. (2004)
Cough with phlegm		2.12 [a]	1.72 – 2.61		
Phlegm		1.33 [a]	0.90 – 1.97		
Wheezing with colds	Coal smoke from cooking and/or heating	1.57	1.07-2.29	Children (7th grade students) in urban, rural Wuhan	Salo et al (2004)
Wheezing w/o colds		1.44	1.05-1.97		
Chronic cough	PM_{10} and SO_2	0.83	0.31 – 2.24	Adult women in rural Anqing.	Venners et al. (2001)
Chronic phlegm		1.52	0.52 – 4.45		
Wheezing		2.91 [c]	1.18 – 7.18		
Shortness of breath		2.87 [b]	1.46 – 5.64		
Wheezing	Wood and hay smoke	1.36 [a]	1.14 – 1.61	Adults >14 in rural Anhui.	Xu et al. (1996)
Wheezing	Coal	1.47 [a]	1.09 – 1.98		

Source: Reproduced from Larsen (2007a). * Some of the studies reported odds ratios instead of relative risk ratios. The difference is however minimal for the prevalence or incidence rates of the health outcomes in the table. (a) No information about significance levels (P-values); (b) Statistically significant at 1%; (c) Statistically significant at 10%.

[12] Note: Desai et al (2004) did not include Ezzati and Kammen (2001) in their meta-analysis because the risk ratios in Ezzati and Kammen did not easily convert to a simple dichotomous outcome (exposed or not exposed).

Table 3.3 Odds ratios of ARI from SFU air pollution exposure

PM 10 (ug/m^3)	Children under 5 years	Age group 5-49 years
<200	1.0	1.0
200-500	2.42	3.01
500-1000	2.15	2.77
1000-2000	4.30	3.79
2000-3500	4.72	-
2000-4000	-	4.49
>3500	6.73	-
4000-7000	-	5.40
>7000	-	7.93

Source: Reproduced from Ezzati and Kammen (2001).

There are very few studies of the economic benefits and costs of interventions to reduce household air pollution from fuel use. Four recent studies are reviewed in this paper. Two of them are global studies estimating costs and benefits at the regional level. The two other studies are from Colombia and Peru.

Mehta and Shahpar (2004) present a cost-effectiveness analysis of household air pollution control interventions by WHO regions with significant SFU prevalence. Benefits are healthy years gained from reduced risk of ALRI in children and COPD in adult females and males based on regional data from WHO. An improved stove is assumed to reduce SFU pollution exposure and health effects by 75 percent. Per household annualized cost of cooking systems range across regions from $40-90 for LPG, $10-20 for kerosene, and $3-24 for improved stove ($3-5 in Africa and Asia). The system cost for LPG and kerosene includes recurrent fuel cost. Program cost of interventions is included, but is a very small fraction of cooking system cost. All values are in purchasing power parity international dollars. Non-health benefits such as fuel savings and/or time savings are not included.

Table 3.4 present the results from Mehta and Shahpar. Healthy years gained are converted here to US $1,000 and US $5,000 per year gained to produce benefit-cost (B/C) ratios. At US $1,000 per year gained, the B/C ratios are greater than 1.0 for improved stoves in Africa and SEAR D and for kerosene in WPRO B.[13] At US $5,000 per year gained, the B/C ratios are > 1 also for improved stoves in SEAR B, for kerosene in all regions, and for LPG in WPRO B.

[13] India is the largest country in SEAR D. China is the largest country in WPRO B.

Table 3.4 Benefit-cost ratios of indoor air pollution control by WHO regions, 2004 study

	US \$1000 per healthy year			US \$5000 per healthy year		
WHO regions	Improved stove	LPG	Kerosene	Improved stove	LPG	Kerosene
Afro D	2.01	0.16	1.00	10.1	0.8	5.0
Afro E	1.38	0.09	0.50	6.9	0.5	2.5
Amro B		0.07	0.41		0.4	2.1
Amro D	0.17	0.13	0.85	0.9	0.7	4.2
Emro D	0.13	0.09	0.56	0.6	0.5	2.8
Sear B	0.85	0.07	0.41	4.2	0.3	2.0
Sear D	1.63	0.14	0.72	8.1	0.7	3.6
Wpro B	0.03	0.71	3.89	0.2	3.5	19.4

Source: Adapted from Mehta and Shahpar (2004). Note: The regions of Amro A, Europe and Emro B are not presented here. SFU is limited in these regions.

Hutton et al. (2006) conducted in collaboration with WHO a global cost-benefit analysis for each WHO region. Benefits are reduced mortality and morbidity for ALRI in children and COPD and lung cancer in adult females and males, time savings from reduced cooking time and fuel collection, time savings from reduced sick days, reduced solid fuel consumption, and local and global environmental benefits.

We discuss the two most promising scenarios presented in Hutton et al, namely an improved stove intervention and use of LPG instead of solid fuel. The improved stove intervention is a chimneyless rocket stove, assumed to reduce SFU health effects by 35 percent. Annualized unit stove cost is US \$ 2.6-3.1. Annualized unit cost of LPG stove and gas cylinder is US \$9-18. LPG consumption is 0.3-0.9 kg per household per day. Annualized program cost per household is less than one dollar in most regions.[14] Reduced mortality is valued at the human capital value (HCV) and morbidity is valued using the cost-of-illness (COI) approach. We therefore convert the health benefits to DALYs for the purposes of this paper.

Time savings from are valued at 100 percent of wages for adults (approximated by GNI per capita) and 50 percent of GNI per capita for children. Rural households are assumed to purchase 25 percent of baseline fuel wood consumption. Urban households are assumed to purchase 75 percent of fuel wood consumption. Solid fuel savings from interventions are treated as a benefit. Local environmental effects are assessed as fewer trees cut down, while global environmental effects considered are lower CO_2 and CH_4 emissions, valued using current market values of emission reductions on the European carbon market.

[14] All annualized costs reflect a discount rate of 3 percent.

Benefit-cost ratios are presented in table 3.5. The benefit-cost ratios do not include time benefit, solid fuel cost savings and environmental benefits in order to make results more comparable across studies. With health benefits valued at US $1,000 per DALY, the B/C ratios for an improved stove are > 1 in the Africa regions, EMRO D (including Pakistan) and SEAR D (including India). The B/C ratios for LPG are > 1 only in Africa. At US $5,000 per DALY, the B/C ratios for an improved stove are > 1 in all regions except AMRO B. For LPG, the B/C ratios are > 1 for Africa and SEAR D.

As observed from table 3.5, the B/C ratios for both interventions are significantly lower in the AMRO regions, SEAR B and WPRO B (including China) than in the other regions. This is because the health benefits per beneficiary population are much lower (lower baseline ALRI mortality in children) than in Africa, EMRO D and SEAR D (table 3.6). The relatively low B/C ratios for WPRO B are also because of the modeling of delayed benefits in terms of COPD which generally develops from long term exposure to SFU smoke.

Mehta and Shahpar (2004) and Hutton et al have similar findings for the improved stove intervention across regions, with Hutton et al finding somewhat higher B/C ratios. For LPG, however, Hutton et al find substantially higher B/C ratios for Africa, both compared to Mehta and Shahpar and to other regions. One important reason for this is the lower household consumption of LPG in Africa (table 3.6).

Table 3.5 Benefit-cost ratios of indoor air pollution control by WHO regions, 2006 study

	US $1000 per DALY		US $5000 per DALY	
WHO regions	Improved stove	LPG	Improved stove	LPG
Afro D	7.45	1.94	37.2	9.71
Afro E	5.00	1.48	25.0	7.38
Amro B	0.02	0.01	0.11	0.06
Amro D	0.42	0.12	2.11	0.59
Emro D	1.66	0.61	8.31	3.03
Sear B	0.45	0.08	2.23	0.39
Sear D	2.10	0.49	10.5	2.47
Wpro B	0.53	0.13	2.65	0.66

Source: Adapted from Hutton et al (2006). Note: The regions of Amro A, Europe and Emro B are not presented here. SFU is limited in these regions.

Table 3.6 Benefits and costs of indoor air pollution control

Interventions and variables	Africa		The Americas		E Med.	S + SE Asia		W Pac
	AFR-D	AFR-E	AMR-B	AMR-D	EMR-D	SEAR-B	SEAR-D	WPR-B
1. Reduce by half those without modern cooking fuel by switching to LPG								
Beneficiary population (million)	185	171	71	18	97	118	378	318
Total deaths avoided (000)	108	91	1.1	1.9	24	10	110	72
DALYs avoided (000)	3,571	3,003	25	55	741	155	2,931	828
Stove cost (US $ annualized per stove)	9.87	9.83	8.98	9.52	17.80	15.78	6.74	15.25
Program cost (US $ annualized per HH)	0.45	0.23	1.26	0.51	0.35	0.22	0.15	0.40
LPG consumption (kg/HH/day)	0.285	0.285	0.880	0.880	0.385	0.405	0.530	0.545
Total cost (US$ million)	1,838	2,033	2,058	465	1,225	1,964	5,928	6,271
B/C ratio (DALY=US$1,000) health only	1.94	1.48	0.01	0.12	0.61	0.08	0.49	0.13
B/C ratio (DALY=US$5,000) health only	9.71	7.38	0.06	0.59	3.03	0.39	2.47	0.66
2. Reduce by half those without improved stove								
Beneficiary population (million)	102	111	111	23	115	78	365	418
Total deaths avoided (000)	38	32	0.4	0.7	9	3.4	38	25
DALYs avoided (000)	1,251	1,049	9	19	259	54	1,025	289
Stove cost (US $ annualized per stove)	3.08	3.07	2.71	2.87	2.93	2.86	2.65	2.76
Program cost (US $ annualized per HH)	1.17	0.72	3.85	1.43	0.90	0.65	0.43	0.02
Total cost (US$ million)	168	210	415	46	156	122	487	546
B/C ratio (DALY=US$1,000) health only	7.45	5.00	0.02	0.42	1.66	0.45	2.10	0.53
B/C ratio (DALY=US$5,000) health only	37.23	24.98	0.11	2.11	8.31	2.23	10.52	2.65

Source: Adapted from Hutton et al (2006). Note: The regions of Amro A, Europe and Emro B are not presented here. SFU is limited in these regions.

Larsen (2005) and Larsen and Strukova (2006) provide an economic analysis of indoor air pollution control interventions in rural Colombia and rural Peru, respectively. Health benefits include ALRI mortality in children, COPD mortality and morbidity in adult women, and ARI morbidity in children and adult women. Relative risk ratios used to estimate health benefits are from Desai et al. (2004). For ARI risk in adult women, the same relative risk as for children is applied. Risk of COPD for adult males is not included. An improved wood stove is assumed to reduce SFU health effects by 50 percent in both countries. LPG eliminates SFU health effects.

Both studies use a similar approach to valuation of health effects. Mortality is valued in two scenarios using a value of statistical life (VSL) and the human capital value (HCV) for adults.[15] Child mortality is valued using HCV (table 3.7). Morbidity is valued using the cost-of-illness (COI) approach. Time benefits are reduced fuel wood collection time, assuming all fuel wood is collected by the households, valued at 75 percent of average rural wage rates. It is assumed that switching to LPG would save the household 30

[15] VSL: Benefit transfer of US $ 2 million (Mrozek and Taylor, 2002) adjusted in proportion to GDP per capita differentials between Colombia and Peru and high income countries.

minutes per day, and using an improved stove would save 10 minutes per day.[16] Cost of an improved wood stove and LPG stove is US $60 each, annualized at 10 percent discount rate. Cost of LPG is market price at typical household energy consumption for cooking. Intervention program cost is US $5 per household.

Table 3.7 Valuation of mortality (US $ per death)

	Colombia	Peru
VSL adults	127,200	148,600
HCV adults	11,300	13,400
HCV children	58,700	68,900

Source: Larsen (2005) and Larsen and Strukova (2006).

In Colombia, the largest monetized health benefits are reductions in ARI morbidity followed by COPD mortality. Reduction in COPD morbidity and ALRI mortality provides the smallest benefit. ALRI mortality in children is the smallest benefit because of the country's relatively low child mortality rate. In Peru, the largest monetized health benefits are ARI morbidity in children and adult women followed by ALRI mortality in children (valued at HCV).

The benefit-cost ratios are highest for installation of an improved wood stove, ranging from 4.3 to 10.5 in Colombia and 5.4 to 9.2 in Peru (tables 3.8-9). In Colombia, switching to LPG from unimproved stove gives a B/C ratio of 1.2 to 3.3, while switching to LPG from an improved stove has a B/C ratio > 1.0 if adult mortality is valued at VSL. In Peru, the B/C ratio is only greater than one for switching to LPG from unimproved stove when adult mortality is valued at value of statistical life (VSL) or time benefits are included. The studies do not present health benefits in DALYs.

[16] The benefits of reduced fuel wood consumption would likely be larger than the assumed value of time benefits for households that purchase some or all of their fuel wood.

Table 3.8 Benefit-cost ratios of indoor air pollution control interventions in rural Colombia

Benefits of interventions	Health Only		Health and time benefits	
Valuation of mortality	HCV for children and adults	HCV for children; VSL for adults	HCV for children and adults	HCV for children; VSL for adults
B/C ratios				
Improved wood stove (from unimproved stove)	4.3	7.8	7.0	10.5
LPG (from unimproved stove)	1.2	2.2	2.3	3.3
LPG (from improved stove)	0.6	1.1	1.4	1.9

Source: Larsen (2005).

Table 3.9 Benefit-cost ratios of indoor air pollution control interventions in rural Peru

Benefits of interventions	Health Only		Health and time benefits	
Valuation of mortality	HCV for children and adults	HCV for children; VSL for adults	HCV for children and adults	HCV for children; VSL for adults
B/C ratios				
Improved wood stove (from unimproved stove)	5.4	6.8	7.8	9.2
LPG (from unimproved stove)	0.8	1.0	1.4	1.6
LPG (from improved stove)	0.4	0.5	0.8	0.9

Source: Larsen and Strukova (2006).

The estimated health benefits of indoor air pollution control in rural Colombia and Peru can be converted to DALYs.[17] We use the same cost of improved wood stove and LPG stove (US$60 per stove), but now annualized over 10 years at 6 percent discount rate instead of a 10 percent rate. Intervention program cost remains US$5 per household.

At DALYs valued at US $1,000, the B/C ratios for improved stoves range from 1.5 to 1.9 while the B/C ratios for LPG are << 1 (table 3.10). At DALYs valued at US $5,000, the B/C ratios for using LPG instead of an unimproved stove is in the range of 1.3-1.9. However, the B/C ratio remains < 1 for using LPG instead of an improved stove. The B/C ratios for LPG are significantly lower in Peru because of the substantially higher cost of LPG in Peru than in Colombia.

The B/C ratios increase significantly when time savings are included as a benefit. At DALYs valued at US $5,000, and time savings at 75 percent of rural wages, the B/C ratios for using LPG instead of an improved stove are greater than 1 in both Colombia

[17] The following conversions per case are used: Child mortality= 34 DALYs; COPD adult mortality=6 DALY; COPD morbidity=2.25 DALYs; ARI morbidity in children under-5=165/100,000 DALYs; and ARI morbidity in female adults=700/100,000 DALYs.

and Peru. However, the B/C ratio is < 1 in Peru if time savings are valued at less than 75 percent.

Intervention program cost and annualized improved wood stove cost is of comparable magnitude. A lower or higher cost of either of these cost components will therefore have a significant effect on the B/C ratios. In the case of substituting to LPG, the intervention program costs and stove costs are only on the order of 10 percent of total cost, with annual cost of LPG being on the order of 90 percent. Changes in the price of LPG will therefore significantly affect the B/C ratios.

Table 3.10 Benefit-cost ratios for indoor air pollution control in rural Colombia and Peru

	Colombia	Peru	Colombia	Peru
	DALY= US $1,000		DALY= US $5,000	
Health benefits (only):				
Improved stove	1.50	1.87	7.50	9.35
LPG (from unimproved stove)	0.38	0.26	1.90	1.31
LPG (from improved stove)	0.19	0.13	0.95	0.65
Health and time savings benefits:				
Improved stove	4.55	4.50	10.55	11.98
LPG (from unimproved stove)	1.54	0.81	3.05	1.86
LPG (from improved stove)	0.96	0.50	1.72	1.02

Source: Benefits converted to DALYs from Larsen (2005) and Larsen and Strukova (2006).

The relative risk of ALRI from household SFU presented in Desai et al. (2004), and used in all the CBA studies, is for morbidity. Only one study of ALRI mortality was identified in their literature review. The relative risks of ALRI morbidity are therefore applied to ALRI mortality to provide national and global estimates of disease burden from SFU. However, case fatality rates from ALRI differ substantially across groups of children within countries. An important question is therefore how may the approach influence a cost-benefit analysis of interventions to control or prevent indoor air pollution and what are potential implications in terms of strategic targeting of interventions.

Nutritional status is an important factor that influences ALRI mortality in children u5. Fishman et al. (2004) estimate that the relative risk of ALRI mortality in moderately and severely underweight children is 4-8 times higher than in non-underweight children from a review of studies in four African and six Asian countries (table 3.11).

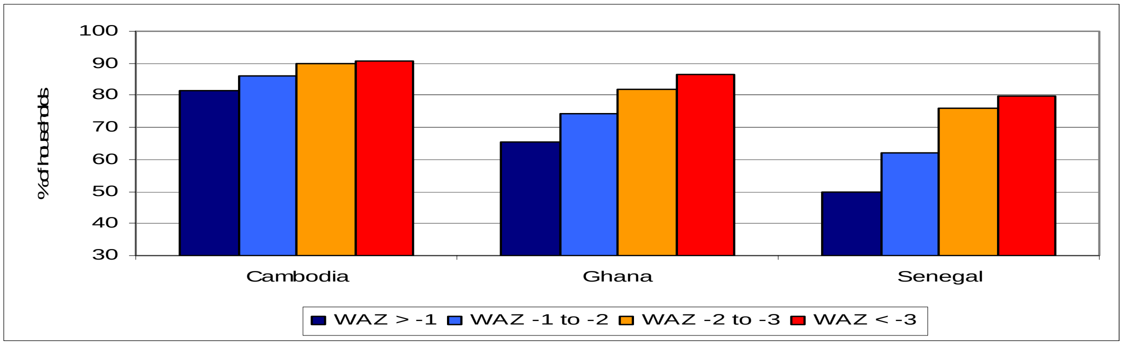
Table 3.11 Relative risk of ALRI mortality by child nutritional status

	Relative risk ratio (RR)
Severe underweight (WAZ < -3)	8.09
Moderate underweight (WAZ -2 to -3)	4.03
Mild underweight (WAZ -1 to -2)	2.01
Non-underweight (WAZ > -1)	1.00

Source: Fishman et al (2004).

We tabulated household fuel use by underweight status of children in the households from the most recent demographic and health surveys (DHS) in Cambodia, Ghana and Senegal. Figure 3.1 presents prevalence of fuel wood use by underweight status in these countries. Use of fuel wood is far more prevalent in households with underweight children, especially in Senegal where the household energy transition has advanced furthest. The use of charcoal in Ghana, often considered an intermediate fuel on the household energy ladder and generally less polluting than fuel wood, is two times more prevalent in households with non-underweight children than in households with severely underweight children. In Cambodia, where households only in the last 5 years have started to switch away from fuel wood, a similar trend is emerging as in Ghana. In Senegal, the use of LPG is nearly 3 times more prevalent in households with non-underweight children than in households with moderately and severely underweight children.

Figure 3.1 Household use of fuel wood by children's underweight status



Source: Cambodia DHS 2005, Ghana DHS 2003 and Senegal DHS 2005. Tabulated by the author.

To illustrate the potential effect of nutritional status on estimation of mortality from SFU and benefit-cost ratios of interventions, we considered a situation typically representative of Sub-Saharan countries where at least 90 percent of households use solid fuels (table 3.12). Around 38 percent of children u5 are mildly underweight and 32 percent are moderately or severely underweight. Only 10 percent of households use LPG, concentrated in households with non-underweight or mild underweight children. Applying the relative risks of ALRI in children in Desai et al and Fishman et al, and substitution to LPG in households with children of different nutrition status, gives benefit-cost ratios presented in figure 3.2. These B/C ratios are relative to a normalized B/C ratio = 1 in the approach that ignores nutritional status. The B/C ratio for severely underweight children is 6 times higher, and the B/C ratio for non-underweight children is 23 percent lower than a B/C ratio that ignores nutritional status.

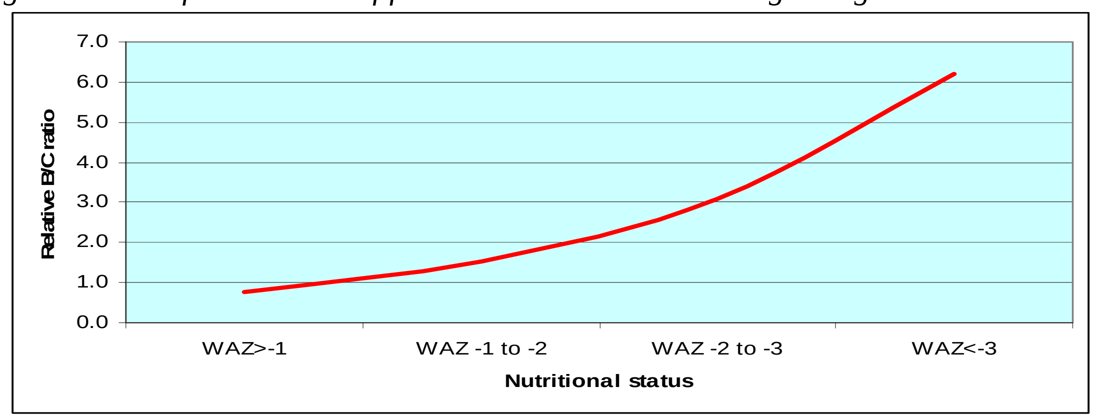
Clearly, a careful analysis of SFU in relation to nutritional status is needed to establish relative risks of ALRI mortality from SFU. However, the simple estimation above may suggest that the B/C ratios for interventions reaching households with high propensity to have malnourished children may be several times higher than the B/C ratios reported in this paper, not only for LPG but also for improved stoves.

Table 3.12 SFU in relation to children's nutritional status in a typical Sub-Saharan country (% of households)

	Weight for age (WA)				
	Non-underweight	Mild underweight	Moderate underweight	Severe underweight	Total
SFU	24%	34%	25%	7%	90%
LPG	6%	4%	0%	0%	10%
Total	30%	38%	25%	7%	100%

Source: Weight-for-age distribution is average for AFR D and E in Fishman et al (2004). SFU is an approximation to regional SFU prevalence.

Figure 3.2 Benefit-Cost ratio of fuel substitution - relative to ignoring nutritional status



Source: Estimated by author.

Benefit-cost ratios from the four studies reviewed are summarized in table 3.13. "Low" is from Mehta and Shahpar (2004), "high" is from Hutton et al (2006), Colombia is from Larsen (2005) and Peru is from Larsen and Strukova (2006). Only health benefits are included in the B/C ratios to make the results more comparable. B/C ratios for improved stove are presented for DALY= US $1,000, and LPG for DALY=US $5,000.

Both global studies find that B/C ratios > 1 for improved stoves in the Africa regions and SEAR D. The B/C ratios < 1 for WPRO B warrant further investigation, as one-third of all mortality from SFU is in this region (especially China). The findings for EMRO D is mixed, with the B/C in Hutton et al being more than 15 times higher than in Mehta and Shahpar.

The low B/C ratios found for the AMRO regions in the two global studies are in contrast to the findings for Colombia and Peru. However the studies differ in important respects. The studies in Colombia and Peru assess benefits and costs in rural areas, where mortality rates are higher than in urban areas. Thus benefits of interventions are estimated to be higher than in the global studies. The Colombia and Peru studies include ARI morbidity for adult women. This is 10-20 percent of DALYs and does therefore not explain the difference with the global studies and do not substantially affect B/C ratios. Further CBA analysis should therefore be undertaken in countries with relatively low child mortality rates, such as in AMRO and SEAR B.

The global studies are inconclusive for LPG, even at US $5,000 per DALY. However, Hutton et al do find B/C ratios >> 1 in the Africa regions, EMRO D and SEAR D which together account for over 60 percent of mortality from SFU. Both global studies assessed benefits and costs of going from currently used stoves to LPG. A majority of these stoves are unimproved in most regions. The B/C ratios may therefore be expected to be lower if an assessment was undertaken for going from improved stoves to LPG. This incremental analysis was done in the Colombia and Peru studies, with the B/C ratios of going from unimproved stove to LPG being twice as high as when going from improved stoves to LPG, e.g., 1.9 versus 0.95 in Colombia.

Table 3.13 Summary of B/C ratios for indoor air pollution control

	Improved stoves		LPG	
	DALY=US $1,000		DALY=US$5,000	
	"low"	"high"	"low"	"high"
Afro D	2.0	7.5	0.8	9.71
Afro E	1.4	5.0	0.5	7.38
Amro B		0.02	0.4	0.06
Colombia (Amro B)	1.5		0.95 - 1.9	
Amro D	0.2	0.4	0.7	0.59
Peru (Amro D)	1.9		0.65 - 1.3	
Emro D	0.1	1.7	0.5	3.03
Sear B	0.9	0.5	0.3	0.39
Sear D	1.6	2.1	0.7	2.47
Wpro B	0.03	0.5	3.5	0.66

Note: Benefits are reduced health effects only. Time and fuel savings would increase the B/C ratios.

None of the studies evaluated the option of replacing fuel wood (and other unimproved biomass fuels) with charcoal. Charcoal is not considered a clean fuel like LPG, but is nevertheless a preferred option for many urban households. Based on a study by Ezzati and Kammen (2001) in Kenya, Gakidou et al (2007) suggests that the relative risk of ALRI from charcoal is 1.3, in contrast to 2.3 from biomass and largely unimproved stoves (Desai et al, 2004). If so, then the health effects of charcoal may be lower than many improved wood stoves. For instance, preliminary results from intervention trials in Guatemala suggest that the excess risk of ALRI is lowered by 50 percent from improved stoves with chimney.

The B/C ratios summarized in table 3.13 are likely to be conservative for many reasons. Country prevalence of SFU and child mortality rates are correlated within a region, such as in AMRO. Thus using regional averages would tend to underestimate the health effects. Important benefits such as time and fuel savings are not included in the B/C ratios. Including these benefits, even at conservative unit values, will substantially increase the ratios.

As already discussed, targeting of interventions to households with high risk of mortality from SFU (e.g., households prone to have malnourished children) may provide substantially greater benefits than a more generic program. If such households are targeted with multiple interventions, a benefit-cost analysis should be undertaken within a multiple risk framework in order to avoid overestimation of benefits of each intervention. Gakidou et al (2007) exemplifies an analysis of targeting high risk households by contrasting a program targeting households with low socioeconomic status with a general program. Recent examples of benefit estimation (or cost of inaction) of environmental interventions (such as controlling indoor air pollution) in a multiple risk framework are Gakidou et al (2007), Larsen (2007b), and a forthcoming report on the health effects and costs of environmental risk factors including indirect effects through malnutrition (World Bank, 2008a forthcoming).

URBAN AIR POLLUTION

1. The Challenge

Particulate matter (PM) is the urban air pollutant that has most consistently been shown to have the largest health effects in studies around the world. It is especially finer particulates, usually measured as PM10 and PM2.5 that have the largest health effects. Ostro (2004) provides a review of studies of PM and health. Exposure to lead (Pb) is also a major concern. Lead has however been eliminated from gasoline in a majority of countries in the world, but other sources of lead remains a localized issue. The focus of this paper is on PM.

PM air pollution originating in the outdoor environment is estimated to contribute as much as 0.6 to 1.4 percent of the burden of disease in developing regions (WHO, 2002). This excludes air pollution caused by major forest fires (e.g. Indonesia in 1997), and serious accidents causing release of organic chemical substances (such as Bhopal, India in 1984) or radioactive pollution (such as Chernobyl in 1986).

Nearly 50 percent of the world's population or 3.2 billion people lived in urban areas in 2006 (World Bank, 2007). As many as 2.3 billion lived in cities with a population over 100 thousand. The World Bank provides estimates of annual average PM10 concentrations in these cities – over 3000 cities in total. WHO estimates a total of 865 thousand deaths in 2002 as a consequence of PM10 in these cities (WHO, 2007).

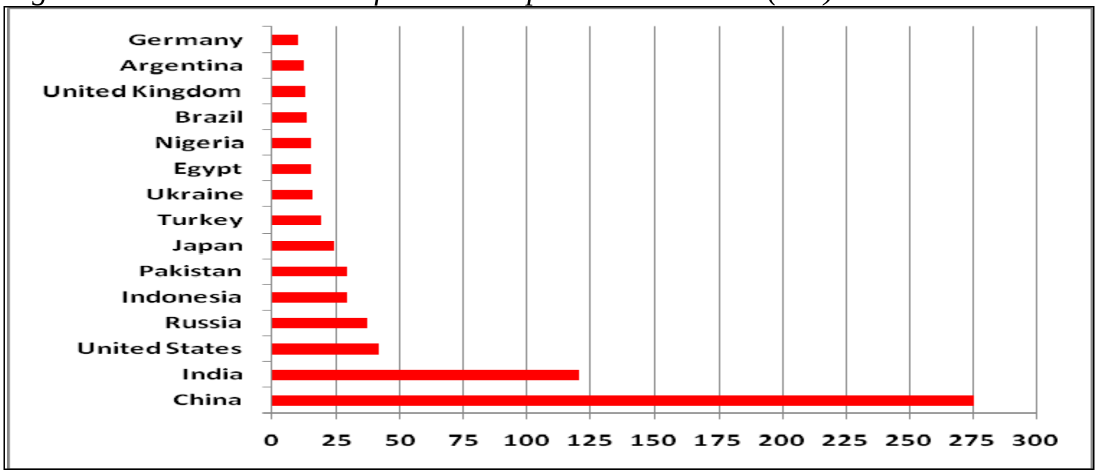
About 85 percent of the deaths from PM in the urban environment occur in low and middle income countries, and more than 55 percent in Asia alone. The death rate from PM is also high in the middle income countries of Europe and Central Asia, because of the high share of elderly and susceptibility to cardiopulmonary disease in this region (table 1.1). Fifteen countries account for 77 percent of global deaths (figure 1.1). China and India alone account for 45 percent of global deaths.

Table 1.1 Estimated deaths from urban particulate matter (PM) in world cities in 2002

	Million Population (In cities > 100,000)	PM10 (Population weighted)	Deaths from PM (thousands)	Deaths from PM (% of total)
East Asia and Pacific*	645	80	327	38%
Europe and Central Asia*	190	36	111	13%
Latin America and Caribbean*	265	43	48	6%
Middle East and North Africa*	125	98	45	5%
South Asia*	250	100	160	18%
Sub-Saharan Africa*	140	71	42	5%
Europe (High income countries)	155	26	56	6%
East Asia and Pacific (High income countries)	145	34	32	4%
North America (High income countries)	240	23	44	5%
Total	2155	60	865	100%

Source: Population and PM10 are from World Bank. Deaths from PM are from WHO (2007). * World Bank regions (low and middle income countries)

Figure 1.1 Estimated deaths from urban particulate matter (PM) in 2002



Source: Adapted from WHO (2007).

Estimated deaths in China of 275 thousand is based on an urban population weighted PM10 of 80 ug/m^3 (WHO, 2007). However, a recent study by the World Bank in collaboration with Chinese institutions estimates deaths from urban PM10 in China at 395 thousand (World Bank, 2008b forthcoming). The study is based on PM10 concentrations in 660 cities with a total population of 580 million, using a log-linear risk function derived from data in Pope et al (2002). PM10 concentration monitoring data used in this study are significantly higher than the PM10 concentrations used by WHO in its estimate of mortality.

There is increasing evidence that it is the very small particulates that cause the greatest health effects. Ambient standards have therefore started to shift to PM2.5. The annual average ambient concentration standard for PM2.5 in the United States is 15 ug/m^3 and WHO has a guideline of 10 ug/m^3. WHO has also tightened its guideline for annual PM10 to 20 ug/m^3 (NAE/NRC, 2008).

Over 55 percent of the urban population in China is exposed to ambient concentrations of PM10 greater than 100 ug/m^3. Ambient PM10 is particularly high in the inland northern half of China, where natural sources contribute substantially to PM concentrations (World Bank, 2008b). Particulate size distribution analysis from Chinese studies indicate that PM2.5 is about half of PM10, thus indicating that over 55 percent of the urban population in China is exposed to PM2.5 exceeding 50 ug/m^3. This implies that about 300 million people in China are exposure to PM2.5 and PM10 ambient levels that are 5 times higher than WHO guidelines.

In the eight largest cities in India, each with a population greater than 3 million and a total population of about 70 million, four cities have annual PM10 ambient concentrations exceeding 100 ug/m^3 (CAI-Asia, 2006). The population weighted average in these cities is over 90 ug/m^3. More than half of the population in these 8 cities is thus exposed to PM2.5 and PM10 levels that are 5 times higher than WHO guidelines.

Table 1.2 presents annual average PM10 ambient concentrations in mega cities in the developing world for which regular monitoring data are available. The cities in Pakistan, Egypt, China, Bangladesh and India have very high PM10 levels. Major cities in Latin America as well as in Thailand and the Philippines have moderate levels of PM10, with the exception of Lima, Peru and Santiago, Chile.

PM10 and PM2.5 concentrations in most Sub-Saharan cities are believed to be moderate. However, available monitoring data from this part of the world are severely limited.

Table 1.2 Annual average PM10 concentration in mega cities in the developing world

City	Country	PM10 (ug/m^3)	Source
Lahore	Pakistan	202	Year 2004 (Krupnick et al 2006 and CAI-Asia 2006)
Karachi	Pakistan	194	Year 2004 (Krupnick et al 2006 and CAI-Asia 2006)
Cairo	Egypt	170	Year 2004-2006 (EEAA 2007)
Beijing	China	145	Years 2003-2004 (CAI-Asia, 2006)
Dhaka	Bangladesh	140	Years 2002-2006 (CAI-Asia, 2006)
Delhi	India	130	Years 2003-2005 (CAI-Asia, 2006)
Tianjin	China	120	Years 2003-2004 (CAI-Asia, 2006)
Kolkata	India	110	Years 2003-2005 (CAI-Asia, 2006)
Lima	Peru	101	Years 2001-2004 (Larsen and Strukova, 2005)
Shanghai	China	100	Years 2003-2004 (CAI-Asia, 2006)
Tehran	Iran	100	Year 2002 (World Bank 2005)
Jakarta	Indonesia	90	Years 2002-2004 (CAI-Asia, 2006)
Santiago	Chile	82	Years 1997-2003 (Cifuentes et al 2005)
Ho Chi Minh City	Vietnam	80	Years 2001-2005 (CAI-Asia, 2006)
Mumbai	India	75	Years 2003-2005 (CAI-Asia, 2006)
Bogota	Colombia	62	Years 2001-2003 (Larsen, 2004)
Mexico City	Mexico	60	Years 1997-2003 (Cifuentes et al 2005)
Bangkok	Thailand	50	Years 2000-2005 (CAI-Asia, 2006)
Sao Paulo	Brazil	49	Years 1997-2003 (Cifuentes et al 2005)
Manila	Philippines	45	Years 2001-2003 (CAI-Asia, 2006)

Source: Author.

The main health effects of fine particulates are cardiopulmonary mortality and respiratory related illness (Ostro, 2004; Pope et al, 2002). It is therefore the older age groups that are most vulnerable to PM exposure. As urbanization continues in the next decades, and the population becomes increasingly old, the health effects of outdoor PM pollution may therefore be expected to increase if PM concentrations levels do not decline significantly.

China's urban population was over 550 million in 2005, or about 40 percent of the total population. The urban population share is expected to reach 60 percent or over 900 million by the year 2030. India's urban population is expected to grow from 285 million in 2001 to 473 million in 2021 and 820 million in 2051 (CAI-Asia, 2006).

Annual population growth in cities with a population over 100 thousand is presented in table 1.3 for select large developing countries. The average growth was 3.2 percent per year during 1990 to 2004 and 2.8 percent from 2000 to 2004. By 2030, if growth slows to half the rate during 2000-2004 by the year 2030, the population in these cities will have grown by 70 percent. Assuming no change in age and cause of death distribution, mortality from PM pollution may increase by the same rate. This may however be a conservative assumption as the population is expected to age significantly over this period of time.

Table 1.3 Annual population growth in cities with population over 100 thousand in select large developing countries

	Annual growth 1990-2004	Annual growth 2000-2004
China	3.6%	3.1%
India	2.5%	2.2%
Indonesia	4.6%	4.1%
Pakistan	3.4%	3.5%
Turkey	2.6%	2.3%
Egypt	1.8%	2.1%
Nigeria	4.8%	4.2%
Brazil	2.3%	2.1%
Iran	2.7%	2.1%
Bangladesh	3.7%	3.5%
Mexico	1.8%	1.4%
Viet Nam	3.3%	2.8%
Weighted average	3.2%	2.8%

Source: Calculated from World Bank PM10 global database.

2. Solutions

Reducing air pollution exposure is largely a technical issue, and includes removing pollution at its source and filtering pollution away from the source. These technical solutions are implemented in a policy environment that makes it illegal to use a polluting substance or process (e.g. bans on leaded gasoline or asbestos), increases the costs of polluting (polluter pays principle), requires the use of pollution control and prevention technologies, mandates maximum allowable pollution loads, or increases information on or encourages best practices with regard to the use of less polluting technologies and substances.

Applying these policies is often more an economic than a technical issue. A cost-benefit analysis of the most effective options can therefore help promote policies for pollution control and prevention. It does however first require identification of the most significant sources of pollution and effective options to reduce pollution from these sources. We therefore start out with a review of so called PM source apportionment studies, PM emission inventories, and projection of future emission from major pollution sources in some of the countries with the highest death toll from outdoor air pollution.

Several PM2.5 source apportionment studies have been conducted in Beijing, representing a city where natural source contribution to PM2.5 ambient concentrations is expected to be significant because of the semi-arid conditions in northern China. The five studies reviewed here find that primary particulates from coal combustion contribute

7-20 percent of ambient PM2.5 concentrations, with a median of 15 percent. The contribution from coal is especially high in the winter. Vehicle emissions contribute 5-7 percent in three of the studies and over 25 percent in Zhang et al (2007). The high contribution found by Zhang et al could be associated with the monitoring sample site being near a major traffic area. Biomass burning contributes 5-15 percent with a mean of about 8 percent. Secondary particulate, mainly from sulfur dioxide and nitrogen oxides, is a major source of ambient PM2.5, with a contribution in the range of 10 to over 30 percent.

Table 2.1 PM2.5 source apportionment studies from Beijing

Method*	CMB	CMB	PMF	PMF	CMB
	Zhang et al (2004)	Zheng et al (2005)	Song et al (2006)	Zhang et al (2007)	Zhang et al (2007)
Coal combustion	16.4%	7%	15.8%	13.8%	20%
Vehicle emissions	5.6%	7%	5.5%	28.5%	25%
Fugitive/road/soil/construction dust	21.4%	20%	7.0%	19.9%	19%
Biomass burning	4.5%	6%	10.1%	11.7%	15%
Industry			4.7%	4.8%	5%
Secondary particulates	9.6%	33%	31.0%	19.2%	15%
Organic matter	15.0%	11%			
Other		2%			
Unexplained	27.5%	14%	25.9%	2.2%	1%
TOTAL	100.0%	100.0%	100.0%	100.0%	100.0%
Average measured PM2.5 (ug/m3)	122	101	93	142	142

Source: Adapted from NAE/NRC (2008), Zheng et al (2005), and Zhang et al (2007). * CMB=chemical mass balance; PMF=positive matrix factorization.

Primary commercial energy consumption in China doubled from 1990 to 2005. The largest share of the increase in energy consumption came from coal followed by petroleum products. Coal consumption was over 65 percent of total primary commercial energy consumption in 2005. Industry including power generation consumed over 70 percent of commercial primary energy in 2005, while transportation consumed less than 10 percent (NAE/NRC, 2008).

As of March 2007 there were 148 million vehicles in China. Some 52 million were 4+ wheelers (of which nearly 13 million were private cars) and 83 million were motorcycles. Car sales rose to over 5 million in 2006 (Reuters, 2007). In 1999, 49 percent of 4+ wheelers were trucks, 20 percent were buses, and 31 percent were cars. Cars are expected to constitute 70 percent of the fleet in 2030 (Blumberg, 2006). In perspective, there were a little over 5 million 4+ wheelers in 1990. This figure is expected to grow to nearly 300 million in year 2035. Motorcycles are expected to grow to 140 million (CAI-Asia, 2006).

With the projected growth in the vehicle fleet in China, vehicle emission contribution to PM2.5 may be expected to grow substantially (especially secondary particulates). Blumberg et al (2006) report that primary and secondary PM2.5 emissions from vehicles are projected to grow seven-fold from 2005 to 2030 in major cities in China if vehicles and fuels only meet Euro 2 standards. Even if vehicles and fuels meet Euro 4 standards, PM2.5 emissions will nearly double.

Chowdhury et al (2007) provides PM2.5 emission source apportionment for three major cities in India, based on site monitoring, particulate analysis and chemical mass balance (CMB). The analysis was conducted for each season. Annual averages for each city and an average for the three cities are presented in table 2.2. Overall, diesel and gasoline combustion contribute the largest share to PM2.5 ambient concentrations (26%) followed by road dust (23%). Biomass burning (solid fuels, waste, etc), secondary particulates and other unidentified sources are also significant. A majority of PM2.5 from diesel and gasoline is from road vehicles. Diesel contributes substantially more than gasoline, although PM2.5 from gasoline (mainly from two- and three-wheelers) is very significant in Kolkata.

Table 2.2 PM2.5 source apportionment studies in 3 major cities in India

	Delhi	Kolkata	Mumbai	Average 3 cities
Diesel	15.9%	20.7%	21.0%	19%
Gasoline	4.9%	13.7%	3.6%	7%
Road dust	24.5%	15.1%	28.9%	23%
Coal	7.1%	8.7%	3.7%	7%
Biomass burning	14.6%	14.6%	13.2%	14%
Secondary PM	15.1%	14.2%	18.3%	16%
Other mass (unidentified sources)	17.9%	12.9%	11.4%	14%
TOTAL	100%	100%	100%	100%

Source: Adapted from Chowdhury et al (2007).

A recent PM emission inventory for urban Pune, India also indicates that vehicles, road dust and burning of biomass for cooking are major contributors to PM2.5 (table 2.3). In Pune, however, brick kilns are the largest identified source of PM2.5. In addition, PM2.5 from rural agricultural burning is as significant as all urban sources combined (Gaffney and Benjamin, 2004). Only a share of these emissions contributes to the urban ambient concentrations of PM2.5.

A recent study by GSSR (2004) indicates that as much as over 80 percent of PM emissions from vehicles in Pune are from two- and three-wheelers (table 2.4).

Motor vehicles in India increased from less than 2 million in 1971 to 67 million in 2003. Two-wheelers accounted for over 70 percent of total vehicles in 2003. Total vehicles are projected to increase at a rate of 10 percent per year (assuming 8 percent GDP growth) and reach over 670 million in 2030, of which two wheelers are expected to constitute 425 million (CAI-Asia, 2006). This represents a nearly 10-fold increase from 2003.

Table 2.3 PM2.5 emission inventory estimate for urban Pune, India

	PM2.5
Road dust	23%
Brick Kilns	30%
On-Road Vehicles	19%
Construction Activities	3%
Household cooking (wood, dung, etc.)	15%
Street Sweeping	1%
Trash Burning	3%
Industry and other	6%
TOTAL	100%

Source: Author's estimate from PM10 emission inventory in Gaffney and Benjamin (2004).

Table 2.4 PM emissions from vehicles in Pune, India

	PM emission shares	Vehicle activity shares
Passenger cars	2%	15%
Two-wheelers	56%	66%
Three-wheelers	33%	18%
Buses	7%	2%
Trucks	2%	
Total	100%	100%

Source: Adapted from GSSR (2004).

In Senegal where transportation is highly dieselized (78 percent diesel and 22 percent gasoline), one may expect that road vehicles contribute significantly to urban air particulate pollution. Larsen (2007c) constructed an emissions inventory of contributions to PM2.5 ambient concentrations in Dakar that includes an estimate of secondary particulates and fugitive emissions (road dust, dust from natural sources, etc). The central estimate indicates that 34 percent of ambient concentrations of PM2.5 is from road vehicles (primary particulates), followed by secondary particulates and fugitive emissions (table 2.5). Household cooking is not a contributor to PM in urban Dakar as LPG is the primary fuel in 95 percent of households.

Road vehicles were also found to be major contributors to ambient PM2.5 in Bogota, Colombia where the contribution from fugitive emissions is less than in Dakar (Larsen, 2005). Contribution from road vehicles is followed by secondary particulates, stationary sources, and fugitive emissions (table 2.6).

Table 2.5 Source contribution to ambient PM2.5 in Dakar, Senegal, 2004

	Contribution to ambient PM2.5 (Central estimate)
Road vehicles	34%
Secondary particulates	23%
Fugitive emissions	22%
Solid waste burning	8%
Cement industry	8%
Power plants	3%
Other industry	<2%

Source: Larsen (2007c).

Table 2.6 Source contribution to ambient PM2.5 in Bogota, Colombia

	Contribution to ambient PM2.5 (Central estimate)
Road vehicles	37%
Secondary particulates	30%
Stationary sources	17%
Fugitive emissions	15%
Forest fires/waste burning	1%

Source: Larsen (2005).

The large share of two-wheelers in the vehicle fleet is limited to Asia and three-wheelers are largely concentrated a South Asia and to some extent in South-East Asia. In the rest of the developing world, the passenger car is the predominant motorized form of transportation. In a study of on-road vehicle distribution in 6 cities in developing countries, passenger cars constitute 72-87 percent of on-road vehicle activity with the exception of Pune, India where two- and three-wheelers is dominant (table 2.7). In Lima and Puno, as much as 25 percent of passenger vehicles are diesel fueled.

Table 2.7 On-road vehicle distribution in 6 cities worldwide

	Passenger vehicles	Two- and three-wheelers	Taxi	Buses	Trucks	Non-motorized	Passenger vehicles (diesel share)
Almaty	82.9%	0.1%	n/d	11.6%	4.7%	0.7%	5.8%
Lima	72.0%	1.0%	3.0%	18.0%	6.0%	0.0%	25.0%
Mexico City	79.0%	1.6%	11.0%	3.5%	5.1%	0.0%	0.6%
Nairobi	87.8%	1.7%	0.5%	3.8%	5.4%	0.8%	8.0%
Pune	12.0%	68.3%	0.3%	1.5%	1.4%	16.5%	25.4%
Santiago	78.9%	1.2%	7.9%	6.7%	5.3%	0.0%	3.1%

Source: Lents et al (2004).

A very large share of vehicles in developing countries have no or minimal emission control technology. This is especially the case for diesel vehicles, which then emit many times more particulates than gasoline vehicles. Globally, the diesel fuel share in transportation was 44 percent and the gasoline share was 56 percent in 2005. This does not include aviation, but is not limited to road vehicle transport (data are not readily available for road transport separately). Of the countries included in table 2.8, transport dieselization ranges from 17 percent in Nigeria to 85 percent in Pakistan and also shows wide variation within each of the regions. In countries with relatively few passenger cars, the dieselization rate is high because of the dominance of trucks and buses with a significant share of diesel combusted by long-haul trucks outside urban areas. However, passenger car dieselization is also relatively high in some countries, such as France, Senegal, and Peru.

Table 2.8 Diesel fuel share in transportation, 2005

Country	Region	Diesel fuel share in transportation
Pakistan	Asia	85%
India	Asia	71%
China	Asia	51%
Indonesia	Asia	44%
Japan	Asia	39%
Thailand	Asia	27%
Senegal	SS Africa	78%
South Africa	SS Africa	39%
Nigeria	SS Africa	17%
France	Europe	74%
Germany	Europe	53%
United Kingdom	Europe	53%
Russia	Europe	34%
Peru	Latin America	74%
Chile	Latin America	61%
Mexico	Latin America	31%
Egypt	Middle East and North Africa	67%
Saudi Arabia	Middle East and North Africa	45%
Morocco	Middle East and North Africa	36%
United States	North America	27%

Source: From IEA. http://www.iea.org/Textbase/stats/prodresult.asp?PRODUCT=Oil.

As we have seen, vehicles are found to be a major source of particulate emissions and contributor to ambient PM in many developing countries. Projected growth in vehicle fleets also indicates that vehicle emissions will continue to be a major source of pollution in the absence of interventions. Vehicle emission control is therefore the focus of our review of cost-benefit studies.

Emission control, broadly speaking, involves measures that reduces emissions per passenger kilometer travelled and reduces overall transport demand. This includes:

- Emission control devices on new and in-use vehicles (e.g. catalytic converters, diesel oxidation catalysts, diesel particulate traps);
- Cleaner fuels (e.g. low sulfur gasoline and diesel, LPG/CNG);
- Inspection and maintenance (I&M) of in-use vehicles;
- Engine modifications (e.g. four stroke vs. two stroke engines for 2&3 wheelers);
- New technology vehicles (e.g. electric, hybrid, fuel cells, solar);
- Smaller, less polluting vehicles and none-polluting transport modes (e.g. bicycling, walking);
- Transport modal shifts from low to high occupancy transportation (e.g. buses, metros, subways and trains);
- Traffic management to improve vehicle flows (e.g. off-street parking, traffic light management, congestions charges, dedicated lanes for high occupancy vehicles);
- Urban planning and reduced commuting to work; and
- Old-vehicle replacement programs (e.g. old high usage vehicles, replacing three-wheelers for newer four wheel taxis).

Most or all of these measures would need to be included in an effective package of interventions to reduce urban pollution from transport. In light of current transport situations in most developing countries, vehicle emission control devices, cleaner fuels, and I&M are essential measures. Controlling emissions from 2 & 3 wheelers is also essential in many Asian countries.

The European Union has implemented progressively more stringent emission limits for road vehicles in the last 15 years. PM emission standards now in effect (Euro 4) for diesel passenger cars and light commercial vehicles are 75-80 percent more stringent than the Euro 1 standards approved in 1993/94, and the Euro 5-6 will be ten times more stringent than the standards today (table 2.9). PM standards for heavy duty diesel engines are > 90 percent more stringent than the Euro 1 standard (table 2.10).

The Euro standards also substantially reduce the maximum limit for NOx in both gasoline and diesel vehicles. A significant share of secondary particulates is from NOx emissions.

Table 2.9 European Union diesel vehicle emission standards for PM (g/km)

		Light commercial vehicles (by weight class 1-3		
	Passenger vehicles	LCV (I)	LCV (2)	LCV (3)
Euro 1 (1992/94)*	0.14	0.14	0.19	0.25
Euro 2** (1996/98)	0.08	0.08	0.12	0.17
Euro 3 (2000/01)	0.05	0.05	0.07	0.1
Euro 4 (2005/06)	0.025	0.025	0.04	0.06
Euro 5 (2009/10)	0.005a	0.005a	0.005a	0.005a
Euro 6 (2014/15)	0.005a	0.005a	0.005a	0.005a

Source: Adapted from www.dieselnet.com. * The earlier year is for passenger vehicles. The later year is for light commercial vehicles. ** Applicable for IDI engines. Slightly less stringent limits apply for DI engines. a - proposed to be changed to 0.003 g/km using the PMP measurement procedure.

Table 2.10 European Union heavy duty diesel engines emission standards for PM (g/kWh)

Tier	Year	PM
Euro I	1992, < 85 kW	0.612
	1992, > 85 kW	0.36
Euro II	1996	0.25
	1998	0.15
Euro III	*1999.10, EEVs only*	*0.02*
	2000	0.1
		0.13*
Euro IV	2005	0.02
Euro V	2008	0.02

Source: Adapted from www.dieselnet.com. * for engines of less than 0.75 dm^3 swept volume per cylinder and a rated power speed of more than 3000/min.

To achieve these emissions limits and for emission control devices to perform and operate properly, the sulfur content in the fuel must be limited. Lowering the sulfur in diesel also has its immediate benefits in terms of PM reductions. Maximum allowable sulfur content in vehicle diesel fuel in the European Union was 2,000 ppm in 1994, lower than the content in diesel used in many developing countries today. The maximum allowable content in gasoline and diesel was 500 ppm in 1996, and is now at 50 ppm. "Sulfur-free" diesel and gasoline fuels (≤ 10 ppm S) were available from 2005, and are mandatory from 2009 (table 2.11).

Table 5.3 Maximum allowable sulfur content in vehicle gasoline and diesel fuel in the European Union

	Year	Max sulfur content (ppm)
Euro 2	1996	500
Euro 3	2000	350
Euro 4	2005	50
Euro 5	2009	10

Source: http://www.dieselnet.com/standards/eu/ld.php.

Similar reductions in sulfur content have been implemented in for instance the United States, Japan and other high income countries. An increasing number of developing countries are also moving to low sulfur diesel and have plans to mandate ultra-low sulfur diesel. South Africa has been moving to 500 ppm sulfur in diesel and is planning to limit the sulfur content to 50 ppm in 2010. Botswana, Lesotho, Namibia and Swaziland are also using 500 ppm sulfur diesel imported from South Africa. Mexico, Bolivia, Chile and metropolitan areas of Brazil are using < 500 ppm sulfur diesel. Many Asian developing countries have mandated 500 ppm diesel fuel for diesel vehicles, including China, India, Malaysia, the Philippines, Thailand, and Vietnam and some of them are moving to 50 ppm sulfur diesel (UNEP, 2006; ADB, 2003).

There are however many countries that continue to use high sulfur diesel. Pakistan, Indonesia, Russia, the Central Asian countries of the former Soviet Union, and many countries in Africa and Latin America are reported to use diesel with a sulfur content > 2,000 ppm. Many African and Middle Eastern countries are even using diesel with a sulfur content > 5,000 ppm (UNEP, 2006).

Technical options to substantially reduce PM emissions from in-use diesel vehicles are available after low and ultra-low sulfur diesel is made available in the market. Either of two technologies are used to retrofit in-use diesel vehicles, i.e., diesel oxidation catalysts (DOC) and diesel particulate filters (DPF). DOCs are effective with low sulfur diesel (500 ppm) and DPFs are effective with ultra-low sulfur diesel (50 ppm). DOCs are found to reduce PM emissions in in-use vehicles by 20-50 percent while DPFs quite consistently reduces PM by over 80-90 percent.

A recent report by UNEP (2006) summarizes the global experience with DOC and DPF. DOCs have been installed on over 50 million diesel passenger vehicles and more than 1.5 million buses and trucks worldwide. DPFs have been installed on over 1 million new diesel passenger vehicles in Europe. From this year, all new on-road diesel vehicles in the United States and Canada are equipped with a high-efficiency DPF. And from 2009, all new diesel cars and vans in the European Union will have to be equipped with DPF. Worldwide, over 200 thousand heavy duty vehicles have already been retrofitted with DPF.

DOCs and DPFs have been used for retrofitting of buses and trucks in many countries on a wider scale or in demonstration projects in Chile (Santiago), China (Beijing), Europe, Hong Kong, India (Pune), Japan, Mexico (Mexico City), Taiwan, Thailand (Bangkok), and United States. These technologies are expected to be increasingly used in developing countries as they move to low and ultra-low sulfur diesel fuel.

3. Economic Estimates of Costs and Benefits

Despite its importance, very few full cost-benefit analyses have been conducted on measures to address outdoor air pollution. Most studies are single country or single city in nature. Hence no global estimates are possible. Existing studies cover several major industrialized countries or economic areas (e.g. USA, Europe, Japan, Canada, and UK) and some heavily polluted cities located in developing countries. Differences that may reduce transferability of results to other settings include different economic levels and valuation of health benefits and different pollution levels and population exposure. In most studies, economic gains measured are limited to reductions in premature deaths, lower health care costs and work days gained due to less morbidity. In only few studies were other economic benefits included, such as avoided damage to agriculture and ecosystems, or avoided damage to infrastructure and public buildings from corrosive pollutants.

Five recent studies of road vehicle emission control in developing countries are discussed here, namely Stevens et al (2005) from Mexico City, Blumberg et al (2006) from China, Larsen (2005) from Bogota, Colombia, ECON (2006) from Lima, Peru, and Larsen (2007c) from Dakar, Senegal. These studies focus on particulate emission control from improved vehicle fuels and control technologies, and represent CBA in high-middle income, low-middle income, and low-income countries in three regions of the developing world.

All the studies use a value of statistical life (VSL) to monetize the benefits of reduction in mortality. Larsen (2005), ECON (2006) and Larsen (2007c) also use a human capital value (HCV). The results from these five studies are discussed and presented here with the VSL approach to provide consistency across the studies. The VSL/GDP per capita ratio ranges from about 66 in the Colombia, Peru and Senegal studies to 74 in China in 2005 and 89 in Mexico.[18] The Colombia, Peru and Senegal use the cost-of-illness (COI) approach for valuation of morbidity. The China study uses a combination of COI and willingness-to-pay (WTP). The Mexico study does not include morbidity benefits of emission reductions.

Older studies, such as Larsen (1994; 1997) from Iran and Morocco and Eskeland (1994) from Chile are not discussed because control cost figures are now more than a decade old.

Stevens et al (2005) evaluates the benefits and costs of in-use vehicle emission control from retrofitting diesel vehicles with particulate control technologies in Mexico City, using 2010 as the year of program implementation. Benefits are limited to reduced mortality from reductions in primary particulates and secondary particulates from hydrocarbon gases. Benefits of reduced morbidity are not included (morbidity is often

[18] In contrast, USEPA uses a VSL that is on the order of 200 times GDP per capita in the United States.

found to represent 20-30 percent of total health benefits, depending on valuation methods used). The analysis assumes that ultra-low sulfur diesel is available by 2010, based on planned completion of phase-in by the year 2009.

Health benefits are estimated using emissions intake fractions and particulate concentration-response coefficients from the international literature. A median intake fraction of 80 per million was applied for primary particulates inside the urban area, and 20 per million for primary particulates outside the urban area of Mexico City. Median concentration response coefficients are from Pope et al (2002) for cardio-pulmonary and lung cancer mortality in adults (> 30 years) and from Woodruff et al (1997) for respiratory deaths in infants. VSL is used for valuation of mortality with a median value of US $660,000.

Benefits and costs of the particulate control technologies are evaluated for three types of vehicles: urban transport buses circulating only within Mexico City; delivery trucks that remain within the city; and long-haul tractor trailers used throughout Mexico. Cost of the retrofit control devices in year 2010 in the study ranges from US $1400-1800 for catalyzed diesel particulate filters (DPF), US $2000-2600 for active regeneration DPF, and US $420-450 for diesel oxidation catalysts (DOC). These figures are estimated from published market prices in 2005, adjusted for high-volume production by 2010. Costs are annualized using a 6 percent discount rate over a period ranging from 4 years for the oldest vehicles to 13 years for new trucks. Operation and maintenance costs (fuel penalty and filter cleaning) range from 20 to 35 percent of annualized capital cost for catalyzed DPF and active regeneration DPF, respectively, with no O&M for DOC.

Benefit-cost ratios range from about 1 to over 25, are higher for older vehicles than newer vehicles, and are higher for buses and trucks than for tractor trailers (the latter vehicles used predominantly outside urban areas). The highest benefit-cost ratios are for DOC on older vehicles. The cost per 1,000 retrofitted vehicles is lowest for DOC but net benefits are higher for DPFs because of their higher particulate reduction efficiency.

Table 3.1 Median benefit-cost ratios for diesel vehicle particulate control retrofit in Mexico City

	Buses	Trucks	Tractor trailers
Older vehicles:			
DPF catalyzed	-	-	-
DPF active regeneration	11.8	5.5	3.1
DOC	27.5	9.4	6.7
Newer vehicles:			
DPF catalyzed	6.1	6.3	2.2
DPF active regeneration	3.5	3.7	1.2
DOC	12.5	7.6	3.0

Source: Calculated from cost per statistical life saved in Stevens et al (2005). Benefit-cost ratios presented here are therefore approximations because benefits and costs reported in Stevens et al are rounded off.
Note: Older vehicles refer to model year 1993 and older. Newer vehicles refer to model year 1994 or newer. Annual vehicle use is 45-62 thousand km for buses, 29-40 thousand km for trucks, and 87-120 thousand km for tractor trailers (the range for each vehicle type reflects age of vehicles).

Blumberg et al (2006) evaluates the benefits and costs of controlling road vehicle emissions in China from improved vehicle standards and low sulfur gasoline and diesel for the period 2008 to 2030. Quantified benefits are mainly reduced health effects from primary and secondary particulate emissions (PM), but also some benefits (2.1 percent of total benefits) from reduced ground level ozone, increased agricultural yields, reductions in material soiling and degradation of antiquities, and improved visibility (based on findings in Europe and the United States). Emissions of primary and secondary particulates (from NOx) are estimated for three scenarios: (a) baseline emissions based on Euro 2 standards and fuels (500 ppm sulfur content); (b) vehicles with Euro 4 standards by 2010 for light duty vehicles and Euro 5 standards by 2012 for heavy duty vehicles; and (c) vehicles with Euro 4 and 5 standards, ultra-low sulfur (50 ppm) gasoline and diesel by 2010, and 10 ppm gasoline and diesel for heavy duty vehicles in 2012.

Blumberg et al present net benefits of emission reductions from 2008 to 2030. Benefit-cost ratios are presented for years from 2015 to 2030. The ratios range from 2-4 in year 2015 to 14-24 in year 2030. Benefits increases by a multiple of 10-14 over this period. Three factors underlie this increase in benefits: a) increase in exposed population; b) increase in emission reductions; and c) increase in the unit values of mortality and morbidity (table 2). Based on the data presented in Blumberg et al, we estimate that the increases in exposed population and emission reductions account for approximately 55-60 percent of the increase in benefits from 2015 to 2030, and that the increase in unit values of mortality and morbidity accounts for approximately 40-45 percent.

Table 3.2 Benefits and costs of vehicle emission control in China

	Improved vehicle standards		Improved fuels (vehicle standards in place)		Improved vehicle standards and fuels	
Year	2015	2030	2015	2030	2015	2030
PM emission reductions (10,000 tons relative to baseline)	10	35	5	20	15	55
Benefits (US$ billion)	8.4	115	4.3	45	13	160
Benefit-cost ratios	4	24	2	14	3	20

Source: Blumberg et al (2006).

Table 3.3 Valuation of health benefits in China study

	Year 2010	Year 2015	Year 2030
Valuation of mortality	160	250	850
Valuation of morbidity	140	190	450

Note: Values are indexed to 100 in year 2005. Values are approximate from charts in Blumberg et al (2006).

The benefits or mortality reductions appear to account for about 80 percent and morbidity reductions for about 20 percent of total benefits.[19] Benefit-cost ratios are therefore particularly sensitive to estimated mortality reduction and valuation of mortality. Health benefits are estimated using emissions intake fractions and particulate concentration-response coefficients from the international literature. The intake fractions are 29 per million for primary PM and 0.64 per million for secondary PM (from NOx emissions) in 59 cities with a 2002 population over 1 million (13-15 percent of China's population) and 4.5 per million for primary PM and 0.09 per million for secondary PM outside these cities. The concentration-response coefficient for mortality is a 0.41 percent increase in all-cause adult mortality (>30 years) per 1 ug/m3 PM taken from Pope et al (2002). A value of statistical life (VSL) is used to monetize mortality benefits of emission reduction. VSL for each year "t" from 2008 to 2030 is given by:

$$VSL_{2005}\ (cGDP_t\ /\ cGDP_{2005})^{\varepsilon} \qquad (3.1)$$

where $cGDP_t$ is GDP per capita in year "t" and ε is the income elasticity of willingness to pay for mortality risk reduction. Blumberg et al uses a VSL of US $127,400 in base year 2005. This is the mean value from several VSL studies in China. An income elasticity of 1.42 is applied to derive VSL for subsequent years. Blumberg et al report that this elasticity is from a VSL study in Chongqing by Wang and Mullahy (2006). So in year 2015, the VSL is about 55 percent higher than in 2010 and 87 percent higher than in 2008, based on projected GDP per capita growth rates. The VSL in 2030 is 8-9 times higher than in 2008. Unit values for morbidity increases at the rate of GDP per capita.

Incremental fuel costs (500 ppm to 50 ppm sulfur fuel) presented in Blumberg et al, based on modeling of the refinery sector in China, are about US $1.3 per barrel for diesel and US $0.8 per barrel of gasoline. Incremental vehicle costs from Euro 2 to Euro 4 vehicles applied in Blumberg et al are US $150, US $400, and US $1500 per light duty gasoline vehicle, light duty diesel vehicle, and heavy duty diesel vehicle, respectively. Incremental cost of a Euro5 heavy duty diesel vehicle is an additional US $1000. A 20 percent reduction in incremental vehicle cost was applied for every doubling of new vehicle sales. A discount rate of 3 percent was applied to annualize incremental vehicle costs over a period of 10 years for light duty vehicles and 15 years for heavy duty vehicles.

While it is of interest to estimate benefits and costs over an extended period of time, it is also of policy relevance to estimate the benefit/cost ratio for the early years of policy implementation. We therefore use the data in Blumberg et al to estimate the benefit-cost ratios for the year 2010 by using the VSL, morbidity unit values, and exposed population for this year. As costs of emission controls are not presented in Blumberg et al for the year 2010, we apply unit control costs and emission reductions in 2015 to arrive at an approximate benefit-cost ratio for the income level in China in 2010. We then get a

[19] We estimate these benefit shares by estimating the reductions in health effects, based on the data presented in Blumberg et al.

benefit-cost ratio of 2.4 for improved vehicle standards and a ratio of 1.2 for improved fuels. Blumberg et al do not present benefit-cost ratios for gasoline and diesel vehicles and fuels separately, and can not be easily derived from the data provided in the report.

Larsen (2007c) evaluates the benefits and costs of lowering the sulfur content in road transport diesel and retrofitting in-use diesel vehicles with particulate control technology in urban Dakar, Senegal. Benefit-cost ratios are presented for lowering of sulfur from > 2,000 ppm to 500 ppm, and from 500 ppm to 50 ppm, as well as for diesel oxidation catalysts (DOC) and diesel particulate filters (DPF) for several sizes of vehicles and annual usage. Benefits are limited to health effects of primary particulate emissions reductions. Benefits of reductions in secondary particulates from reduction in gaseous emissions are not included. Health benefits are estimated from modeled improvements in ambient air quality and concentration-health response coefficients from the international literature, of which cardio-pulmonary and lung cancer mortality coefficients from Pope et al (2002) are the most significant in terms of total health benefits.[20] Reduction in mortality accounts for 75 percent of total health benefits, based on a VSL of US $45,000 in year 2004.

Benefits and costs of lower sulfur diesel are evaluated for light diesel vehicles and for diesel buses and trucks primarily used within urban Greater Dakar. Incremental cost of 500 ppm sulfur diesel range from US $1-3 per barrel. Incremental cost of 50 ppm sulfur diesel (relative to 500 ppm diesel range from US $2.1-3 per barrel. Benefit-cost ratios range from 1.1 to 3.7 for 500 ppm sulfur diesel and 1.4 to 2.4 for 50 ppm diesel, depending on assumptions of incremental cost of lower sulfur diesel.

Benefits and costs of DOC and DPF for taxis and small and large buses used within urban Greater Dakar are evaluated for a range of annual vehicle usage and useful lifetime of the particulate control devices. Cost of DOC in the analysis is US $1,000 per vehicle. Cost of DPF is US $850 for taxies and US $5,000 for buses. Costs are annualized by using a discount rate of 10 percent over the useful life of the devices, ranging from 5 to 10 years in the analysis. Benefit-cost ratios are > 1 for DOC on high usage buses and taxis, and on low usage buses if the useful life of the DOC approaches 10 years. Benefit-cost ratios are > 1 for DPF for high usage taxis, and for high usage buses if the useful life of the DPF approaches 10 years.

The cost of DOC and DPF applied in this study is about twice the cost applied in Stevens et al (2006). If cost reductions in the next few years are as assumed in Stevens et al, the benefit-cost ratios in Larsen (2007c) would be twice higher than presented here.

[20] Improvements in ambient air quality from emission reductions are estimated based on the development of an all-source particulate emission inventory including area-wide sources and particulates from natural sources.

Table 3.4 Benefit-cost ratios of reducing sulfur in vehicle diesel fuel in Dakar

Diesel (sulfur 500 ppm)	Light diesel vehicles	Diesel buses and trucks
Low cost (US $ 1.0 per barrel)	3.2	3.7
Medium cost (US $ 1.6 per barrel)	2.0	2.3
High cost (US $ 3.0 per barrel)	1.1	1.3
Diesel (sulfur 50 ppm)		
Low cost (US $ 2.1 per barrel)	2.0	2.4
Medium cost (US $ 2.8 per barrel)	1.5	1.8
High cost (US $ 3.0 per barrel)	1.4	1.7

Source: Larsen (2007c). Notes: Light diesel vehicles used 90% in Greater Dakar. Diesel buses and trucks used 100% in Greater Dakar.

Table 3.5 Benefit-cost ratios for in-use diesel vehicle retrofit particulate control in Dakar

	DOC		**DPF**	
	Low usage Vehicles*	High usage vehicles*	Low usage Vehicles*	High usage vehicles*
5 year useful life				
Buses (large)	0.89	1.77	0.38	0.76
Buses (small)	0.55	1.11	0.24	0.47
Taxis	-	-	0.67	1.34
10 year useful life				
Buses (large)	1.43	2.86	0.61	1.23
Buses (small)	0.89	1.79	0.38	0.77
Taxis	-	-	1.08	2.17

Source: Larsen (2007c). * Low and high usage refers to 35,000 km and 70,000 km per year, respectively.

Larsen (2005) evaluates the benefits of lowering the sulfur content in road transport diesel from 1000 ppm to 500 ppm and retrofitting in-use diesel vehicles with particulate control technology in Bogota, Colombia. Benefits are limited to health effects of primary particulate emissions reductions. Benefits of reductions in secondary particulates from reduction in gaseous emissions are not included. Health benefits are estimated from modeled improvements in ambient air quality and concentration-health response coefficients from the international literature, of which cardio-pulmonary and lung cancer mortality coefficients from Pope et al (2002) are the most significant in terms of total health benefits.[21] The study applies a VSL of US $127,200.

The study does not provide benefit-cost ratios, but can be derived by applying unit cost figures. We here apply the central estimate of cost in Larsen (2007c) for Dakar, Senegal. At an incremental cost of US $1.6 per barrel for 500 ppm sulfur diesel, the benefit-cost ratios range from 1.8 to 2.2 for light and heavy duty diesel vehicles. At a cost of US $5,000 for particulate control technology for heavy vehicles, the benefit-cost ratios for

[21] Improvements in ambient air quality from emission reductions are estimated based on the development of an all-source particulate emission inventory including area-wide sources and particulates from natural sources.

diesel buses are in the range of 3-5, and as high as 10-20 for heavy duty diesel trucks. These ratios are for vehicles used within Bogota. The ratios would be substantially lower for inter-urban vehicle use. The difference in benefit-cost ratios for retrofit particulate control technology on buses and heavy duty trucks stems from differences in baseline emissions, annual usage, and emission reductions.

Table 3.6 Benefit-cost ratios for low sulfur diesel and particulate control technology in Bogota

		Particulate control technology	
	500 ppm diesel	5 year useful life	10 year useful life
Light duty diesel vehicles	1.8	-	-
Heavy duty diesel trucks	2.2	10-13	16-20
Diesel buses	1.9	3	5

Source: Estimated based on benefit calculations in Larsen (2005). Note: Annual usage is 50,000 km per year for buses and 66,000 km per year for heavy trucks.

ECON (2006) evaluates a range of options to control particulate emissions from vehicles in Lima, Peru. Benefits and costs are monetized for lowering the sulfur content in diesel from > 2500 ppm to 50 ppm, retrofit particulate control for urban diesel buses, an inspection and maintenance (I&M) program for diesel vehicles, and introduction of new buses using compressed natural gas (CNG) instead of new diesel buses. Benefits are health effects from reductions in primary and secondary particulates.

Improvements in ambient air quality from emission reductions are modeled following the approach in Larsen (2005). Health benefits from these improvements in ambient air quality are estimated based on Larsen and Strukova (2005), which use concentration-health response coefficients from the international literature of which cardio-pulmonary and lung cancer mortality coefficients from Pope et al (2002) are the most significant in terms of total health benefits. The study applies a VSL of US $148,600.

The benefit-cost ratios for ultra-low sulfur diesel (50 ppm) are in the range of 1.3-1.9, based on assessment of the cost of refinery upgrading in Peru using discount rates of 6 and 3 percent to annualize capital cost. A 20 percent reduction in primary particulate emissions is applied, which may be on the low side in light of the high sulfur in diesel in Peru (> 2500 ppm) at the time of the study. The benefit-cost ratios for retrofitting diesel buses with particulate control technologies are in the range of 2.9-5.7, using discount rates of 6 percent over 5 years of useful equipment life and 3 percent over 10 years of useful equipment life. The equipment cost is assumed to be US $3,000 per vehicle, with a particulate emission reduction efficiency of 90 percent. The benefit-cost ratio for an I&M program is 5.4, based on an estimated cost of US $4100 per ton of PM reduction. The benefit-cost ratio for introducing CNG buses instead of diesel buses is found to be significantly < 1 because of the high incremental cost of buses and investment requirements in refueling stations.

Table 3.7 Benefit-cost ratios for vehicle particulate emission controls in Lima, Peru

	B/C ratios	Discount rate
50 ppm diesel (from > 2500 ppm)	1.88	3%
	1.29	6%
Retrofit particulate control for diesel buses	5.66	3%
	2.87	6%
I&M program for diesel vehicles	5.36	
New CNG buses (compared to diesel buses using 50 ppm sulfur diesel)	0.44	3%
New CNG buses (compared to diesel buses high sulfur diesel)	0.71	3%

Source: Adapted from ECON (2006).

Some of the differences in benefit-cost ratios found in the five studies reviewed are due to variation in the VSL used for valuation of mortality benefits. The benefits can however be expressed as DALYs valued in monetary units. The following equation is used to convert benefit-cost ratios using VSL for valuation of mortality benefits (B/C_{VSL}) to benefit-cost ratios with benefits expressed in DALYs valued in monetary units (B/C_{DALY}):

$$B/C_{DALY} = B/C_{VSL} * \beta * YLL * DALY\$/[VSL*(1-\alpha)] \qquad (3.2)$$

where β is share of mortality benefits to total health benefits (mortality and morbidity); YLL is years of life lost per death; DALY$ is US $1,000 or US $5,000; VSL is the value of statistical life applied in each study; and α is the share of life lost to disability (YLD) to total DALYs.

We reviewed Colombia, Peru and Senegal studies conducted for the World Bank that were used to estimate benefits of emission reductions in Larsen (2005), ECON (2006) and Larsen (2007c) and found a β of about 0.75 and an α ranging from 0.43 in Senegal to 0.5 in Colombia and 0.56 in Peru. Using the data presented in Blumberg et al (2006), we also find a β of approximately 0.75. By applying DALY factors (DALYs per case of mortality and morbidity) from the Colombia, Peru and Senegal studies to the estimated health effects in Blumberg et al (2006), we find an α of 0.45 for China. For Mexico, we used β=0.75 and α=0.5. We used YLL=8 in equation 3.2. This reflects age-weighted years of life lost to premature death, discounted at 3 percent per year (the age-weighting and discounting is the standard calculation procedure of DALYs by the World Health Organization).

A summary of the most promising control measures in the five studies indicates that the B/C ratios are highest in Mexico when using VSL for valuation of mortality benefits. This is as expected given that the VSL applied in the Mexico study is 4-5 times higher than the value applied in China, Colombia and Peru, and more than 10 times higher than the value used in the Senegal study.

When health benefits are converted to DALYs and valued at US $1,000 per DALY, the B/C ratios are consistently < 1 and mostly < 0.5.

At US $5,000 per DALY, the B/C ratios for low (500 ppm) and ultra-low (50 ppm) sulfur diesel for vehicles used primarily in urban areas are > 2 in Senegal. The ratios are < 1 in Colombia and Peru primarily because of the relatively low emission reductions assumed in those two studies. The China study uses emission intake fractions (health effects are in most studies assumed to be proportional to the intake fractions) that are much lower than used in the Mexico study and found in three major cities in China according to Blumberg et al (2006). So if similar intake fractions were used in the China study as in the Mexico study, the B/C ratio would be > 1 when DALYs are valued at US $5,000.

At US $5,000 per DALY, particulate control technologies are generally found to have a B/C ratio > 1 for high-usage buses, trucks and taxis used in urban areas. I&M for diesel vehicles, although only evaluated in one of the studies, is also found to have a high B/C ratio.

Table 3.8 Summary of B/C ratios for vehicle particulate emission control

Country	Location	Intervention	B/C ratio		
			VSL	US$1000 Per DALY	US$5000 Per DALY
China	59 cities	Euro 4, 5 vehicle technology	2.4	0.21	1.03
		50 ppm gasoline and diesel	1.2	0.10	0.51
Colombia	Bogota	500 ppm diesel	1.9	0.18	0.90
		DPF for buses	5	0.47	2.36
Mexico	Mexico City	DOC (older buses)	27.5	0.67	3.33
		DOC (newer buses)	12.5	0.30	1.52
		DOC (older city delivery trucks)	9.4	0.23	1.14
		DOC (newer city delivery trucks)	7.6	0.18	0.92
		DPF active regeneration (older buses)	11.8	0.29	1.43
		DPF catalyzed (newer buses and city delivery trucks)	6.2	0.15	0.75
Peru	Peru	50 ppm diesel	1.9	0.18	0.88
	Lima	Retrofit PM control for buses	5.6	0.52	2.59
		I&M program for diesel vehicles	5.3	0.49	2.45
Senegal	Dakar	500 ppm diesel	2.1	0.49	2.46
		50 ppm diesel	1.7	0.40	1.99
		DOC for large buses (high usage; 10 yr life)	2.8	0.65	3.27
		DPF for taxis (high usage; 10 yr life)	2.2	0.51	2.57

There are also benefit-cost analysis studies in low and middle income countries that look at pollution control in the industrial and power sector, fuel substitution, and energy efficiency. Some of the most recent studies are presented in table 3.9. Benefit-cost ratios are mostly in a range from less than one (lower bound) to 6, but are as high as over 100 in the case of Mexico which in part is related to the applied VSL. Some examples of benefit-cost analysis studies in high income countries are presented in table 3.10.

Table 3.9 Examples of cost-benefit studies of outdoor air pollution control in low and middle income countries

Study	Netalieva et al (2005)	Mao et al (2005)	Li et al (2004)	Blackman et al (2000)	Aunan et al (1998)
Location	Kazakhstan	China (2 cities)	Shanghai, China	Ciuded Juarez, Mexico	Hungary
Interventions	Emission reductions in the oil extraction industry	Beijing (B); Chongqing C) substituting natural as for coal	Emissions control: (C1) power; (C2) industrial	PM emissions control from traditional wood fired brick kilns: A: use natural gas; B: improved kilns; C: relocation	Air pollution control in various sectors: Agriculture (A); Industry (B); Transportation; energy (C); Households (D); Services (E)
Benefits	Health	Health	Health, labor productivity	Health	Health
B/C ratios	5.7	B: 29% IRR C: 75% IRR	C1: 1.1 (0.5-2.9) C2: 2.8 (1.3-7.6)	A: 75 B: 107 C: 30	A: 3; B: 5; C: 6; D: 16; E: 17

B/C ratios – benefit cost ratios; IRR – internal rate of return.

Table 3.10 Examples of cost-benefit studies of outdoor air pollution control in high-income countries

Study	US Federal Regulations (USOBM, 2005)	United States (USEPA, 1999)	European Commission (Pye and Watkiss, 2005)	UK Air Quality Strategy review (UKDEFRA, 2006)	Canada (Pandey and Nathwani, 2003)	Japan (Voorhees, 2000)	Japan (Kochi et al., 2001)
Location	US-wide	US-wide	Europe-wide	UK-wide	Canada	Tokyo	Japan-wide
Policies	National emissions standards for hazardous air pollutants	Clean Air Act 5 categories	Air quality targets for CO/Benzene, heavy metals, ozone, hydrocarbons	17 policies to achieve AQS: (reported here: meeting European standards, low and high intensity)	Pollution control program	NOx emission control	SO_2 emissions control: 1: 1968-73 2: 1974-1983 3: 1984-93
Period	1994-2004	1990-2010	NA	Until 2020		1973-1993	1968-1993
Discount rate	7%	5%	2%-6%	HM Treasury rate			2.5%
Costs	Compliance & monitoring	R&D, capital, O&M	NA	Capital and recurrent			Capital, fuel conversion; running
Benefits	Health	Health, crop damage, visibility	Health; labor productivity	Health (1%, 3% and 6% hazard rates reported; optimistic 6% reported here)			Medical expenses, labor losses avoided, adjusted by WTP factor
B/C ratio	2.72-13.0	3.8	6.0	L: 1.5-3.8 H: 0.9-2.3	3.0	6.0	1. 5.39 2. 1.18 3. 0.41

NA – not available; B/C ratio – benefit cost ratio; R&D – research and development; O&M – operations and maintenance; CO – carbon monoxide; AQS – air quality standards; NOx – nitrogen oxides; SO_2 – sulfur dioxide; WTP – willingness to pay.

IMPLICATIONS AND OUTLOOK

In interpreting the results of the reviews in this paper, it is important to keep in mind the multiple uncertainties of cost-benefit analysis in the field of indoor and outdoor air pollution. First, measuring the impact of air pollution on health is very complex since there are many different pollutants and their effects on health are difficult to discern. Hence, controlled trials in the medical scientific sense are very few in the area of air pollution. Second, there exists significant uncertainty as to the improvement in ambient air quality and population exposure from individual interventions. Third, there is uncertainty as to the number of years of life saved from reduced adult mortality risk associated with improved air quality. Different methods and values used between different studies thus makes it difficult to compare the results of studies reported in the literature.

While the control strategies for indoor and outdoor air quality improvement are largely unrelated, they share some similar basic approaches: (1) fuel switching, (2) technology emission control, and (3) fuel use efficiency. Each of these options offers different opportunities and drawbacks. General constraints to the implementation of air pollution control measures cover lack of political motivation or competing political priorities, lack of economic (purchasing) power, lack of regulatory framework or regulation monitoring, and lack of access of the potential user to the necessary resources or technologies. And, importantly, households and the public in general may not be fully aware of the health effects of air pollution, thus effective demand for solutions is not present. Hence, government and private sector activities should focus on addressing these barriers, according to their importance in each air pollution context.

A summary of interventions with the highest B/C ratios are presented in table 1.1. The B/C ratios for improved household cooking stoves are from Hutton et al (2006) for the regions of Africa, EMRO D (incl. Pakistan, Afghanistan) and SEAR D (incl. India, Bangladesh). These are the regions with the largest number of estimated deaths from indoor air pollution, or 65 percent of global deaths. Full adoption of improved household cooking stoves in these regions could potentially save 340-680 thousand lives per year, assuming a 35-70 percent reduction in health effects from the use of improved stoves.

The B/C ratio of improved stoves in the WPR B (incl. China) is well below one for DALYs valued at US $1,000, according to Hutton et al when only health benefits are accounted for. Twenty-seven percent of global deaths from indoor air pollution are in this region, mainly arising from chronic obstructive pulmonary disease (COPD) in adults in China. Further benefit-cost analysis of interventions to control indoor air pollution is therefore warranted.

Three interventions to control urban air pollution from road vehicles are presented in table 1.1. The B/C ratios reflect the range for the technologies assessed in the studies reviewed, and are all below one for DALYs valued at US $1,000, but in the range of 0.9-3.3 for DALYs valued at US $5,000.

In light of the PM apportionment and PM emission inventory studies reviewed here, the interventions in table 1.1 may reduce global health effects of urban air pollution by 10-20 percent. This would correspond to saving 80-160 thousand lives per year. In Asia, controlling pollution from two- and three-wheelers would also be an essential ingredient of improving urban air quality.

Table 1.1 Summary of intervention B/C ratios

Interventions	B/C ratios (DALY= US $1,000)	B/C ratios (DALY= US $5,000)	Annual Benefits (Reduction in mortality, '000 lives)
Improved cooking stoves	1.7 - 7.5	8 - 37	340-680
Low and ultra-low sulfur diesel for urban road vehicles	0.2 - 0.5	0.9 - 2.5	80-160
Diesel vehicle particulate control technology	0.2 - 0.7	0.9 - 3.3	
I&M program for diesel vehicles	0.5	2.5	

The findings from the studies reviewed here raise important questions as to valuation of the health benefits of indoor and outdoor air pollution control. Using a uniform valuation, such as cut-off point per DALY, might be a useful tool to inform the international community of what programs and in which countries can development assistance provide the largest benefits relative to costs. However, the population in the countries are themselves likely to bear most of the costs, such as the cost of improved stoves or LPG, and low sulfur diesel and particulate control technology on vehicles. Individual countries may therefore be interested in knowing more about the benefits of these programs as perceived and experienced by its population, both in terms of child and adult health effects. In this respect VSL is generally believed to better reflect individuals' valuation of mortality risk reduction than a relatively arbitrary value of a DALY. Monetized benefits will therefore to a significant extent vary across countries in relation to per capita income levels. More VSL studies are therefore needed in developing countries, at least to provide reasonably reliable benefit transfers to guide policy makers.

A consideration that is often ignored or inadequately considered in benefit-cost analysis is the cost of programs or incentives to achieve impact and change on a large scale. Achieving adoption and sustained use and proper maintenance of improved stoves or switching to cleaner fuels for indoor air pollution control for a majority of the population is likely to require substantial awareness and promotion programs and follow ups. The studies reviewed in this paper assume a constant cost per household. It may however be that marginal costs are increasing of such programs to achieve demand for pollution control and full population coverage. Limited evidence from hand washing promotion programs suggests a population response rate of 10-20 percent (Borghi et al, 2002; Saade et al, 2001; Pinfold and Horan, 1996). The program cost data in these studies do seem to indicate a potentially rising marginal cost curve. For outdoor air pollution control the situation is somewhat different. Most importantly, governments need to be willing and have the capacity to implement and enforce regulations. This however does also require stakeholder participation (e.g., oil refineries, bus operators, consumers), which is more likely with public and stakeholder awareness of health effects and public demand for cleaner air, as in the end, consumers will bear most of the cost of air pollution control.

When evaluating the benefits and costs of controlling indoor and outdoor air pollution, several important linkages to other areas of the environment and health may be considered. This includes greenhouse gas emission implications of fuel and technology choices. And, when assessing the health benefits of air pollution interventions, it would be preferred to do so in a multiple risk framework to account for potentially simultaneous interventions to improve health. This would help avoid overestimation of health benefits as attributable fractions of disease burden estimated from single risk factors are generally not additive. This issue is particularly relevant for indoor air pollution interventions in relation to interventions to improve child nutritional status, disease treatment and case management, as well as to improved water, sanitation and hygiene. In terms of outdoor air pollution, Martins et al (2004) find that the health effects of urban air pollution in Sao Paulo, Brazil are largest in the lowest socioeconomic groups (several times higher than in the highest socioeconomic groups). Thus addressing effect modifiers may lower the health effects of air pollution.

Potentially important is also targeting of interventions for population groups most vulnerable to health effects and mortality from air pollution. For instance, Gakidou et al (2007) estimate the global health benefits of targeting "poor" versus "rich" households in the developing world in terms of improving child nutrition and providing clean water, sanitation and fuels.

References

ADB, 2003: Cleaner fuels – policy guidelines for reducing vehicle emissions in Asia. Asian Development Bank.

Albalak, R., Bruce, N., McCracken, J., Smith, K., and De Gallard, T. 2001: Indoor Respirable Particulate Matter Concentrations from an Open Fire, Improved Cook Stove, and LPG/Open Fire Combination in a Rural Guatemalan Community. Environmental Sciences and Technology. Vol. 35, No. 13.

Aunan K, Patzay G, Asbjorn Aaheim H and Martin Seip H., 1998: Health and environmental benefits from air pollution reductions in Hungary. The Science of the Total Environment. 212: p. 245-68.

Baranzini, A. and Goldenberg, J., 1996: Desertification, energy consumption, and liquefied petroleum gas use, with an emphasis on Africa. Energy for Sustainable Development. Vol. II, No. 5. January 1996.

Blackman A, Newbold S, Shih J and Cook J., 2000: The benefits and costs of informal sector pollution control: Mexican brick kilns. Resources for the Future, Washington, DC.

Blumberg, K., He, K., Zhou, Y., Liu, H., and Yamaguchi, N., 2006: Costs and benefits of reduced sulfur in China. The International Council on Clean Transportation. December 2006. www.theicct.org.

Borghi, J., L. Guinness, J.Ouedraogo, and V. Curtis, 2002:. Is hygiene promotion cost-effective? A case study in Burkino Faso. Tropical Medicine and International Health, 7(11).

Bruce N, Rehfuess E, Mehta S, Hutton G and Smith K., 2005: Indoor Air Pollution, in Disease Control Priorities in Developing Countries, Jamison D, Breman J, Measham A, Alleyne G, Claeson M, Evans D, Jha P, Mills A and Musgrove P, Editors. 2005, Oxford University Press: New York.

CAI-Asia, 2006: Country synthesis report on urban air quality management. Individual reports for Bangladesh, People's Republic of China, India, Indonesia, Pakistan, Philippines, and Vietnam. Asian Development Bank and the Clean Air Initiative for Asian Cities Center. Discussion draft, December 2006.

Cifuentes, L., A. Krupnick, R. O'Ryan, and M. Toman, 2005: Urban air quality and human health in Latin America and the Caribbean. Paper prepared for the Inter-American Development Bank. Washington DC.

Desai, M., Mehta, S., Smith, K., 2004: Indoor smoke from solid fuels - assessing the environmental burden of disease at national and local levels. Environmental Burden of Disease Series, No. 4. WHO.

ECON, 2006: Urban air pollution control in Peru. Prepared for the Peru Environmental Analysis, World Bank. ECON Analysis, Oslo, Norway.

Edwards, RD., Liu, Y., He, G., Yin, Z., Sinton, J., Peabody, J., Smith KR., 2006: Household CO and PM measured as part of a review of China's national improved stove program. Indoor Air.

EEAA, 2007: Egypt state of environment report 2006. Egyptian Environmental Affairs Agency. Ministry of State of Environmental Affairs. Arab Republic of Egypt.

Eskeland, G., 1994: The net benefits of an air pollution control scenario for Santiago. In: Managing Environmental Problems: Economic Analysis of Selected Issues. World Bank. Report No. 13061-CH.

Ezzati, M. and Kammen, D., 2001: Quantifying the effects of exposure to indoor air pollution from biomass combustion on acute respiratory infections in developing countries. Environmental Health Perspectives, 109(5): 481-488.

Ezzati, M. and Kammen, D, 2002: The health impacts of exposure to indoor air pollution from solid fuels in developing countries: Knowledge, gaps, and data needs. Discussion Paper No. 02-24. Resources For the Future. Washington DC.

Fishman, M.S., Caulfield, L.E., De Onis, M., Blossner, M., Hyder, A.A., Mullany, L., and Black, R.E., 2004: Childhood and maternal underweight. In Ezzati, M., Lopez, A.D., Rodgers, A., and Murray, C.J.L. (Eds): Comparative quantification of health risks – global and regional burden of disease attributable to selected major risk factors. Vol. 1. World Health Organization.

Gaffney, P. and M. Benjamin, 2004: Pune, India regional emissions inventory study. USEPA and Ministry of Environment and Forests, India.

Gakidou, E., S. Oza, C. Fuertes, A. Li, D. Lee, A. Sousa, M. Hogan, S. Vander Hoorn, and M. Ezzati, 2007: Improving child survival through environmental and nutritional interventions – the importance of targeting interventions toward the poor. Journal of the American Medical Association, 298(16): 1876-87.

GSSR, 2004: Pune vehicle activity study. Report by Global Sustainable Systems Research. Updated February 16, 2004. See International Susstainable Systems Research Center. www.issrc.org.

Hutton G, Rehfuess E, Tediosi F and Weiss S., 2006: Global cost-benefit analysis of household energy and health interventions. Department for the Protection of the Human Environment, World Health Organization.

Jin, Y. L., Zhou Z., He G. L., Wei H. Z., Liu J., Liu F., Tang N., Ying B., Liu Y. C., Hu G., Wang H., Balakrishnan K., Watson K., Baris E. and Ezzati M. 2005: Geographical, Spatial, and Temporal Distributions of Multiple Indoor Air Pollutants in Four Chinese Provinces. Environmental Science & Technology 2005; 39(24). 9431-9439.

Kjellstrom T, Lodh M, McMichael T, Ranmuthugala G, Shrestha R and Kingsland S., 2006: Air and water pollution: burden and strategies for control, in Disease Control Priorities in Developing Countries, Jamison D, Breman J, Measham A, Alleyne G, Claeson M, Evans D, Jha P, Mills A and Musgrove P, Editors. 2006, Oxford University Press: New York.

Kochi I, Matsuoka S, Memon MA and Shirakawa H., 2001: Cost benefit analysis of the sulfur dioxide emissions control policy in Japan. Environmental Economics and Policy Studies. 4: p. 219-33.

Krupnick, A., B. Larsen and E. Strukova, 2006: Cost of environmental degradation in Pakistan – an analysis of physical and monetary losses in environmental health and natural resources. Report prepared for the World Bank.

Larsen, B., 2007a: Cost of indoor air pollution in China – rural estimates. East Asia Department Paper (forthcoming). World Bank. Washington DC.

Larsen, B., 2007b: Costs of environmental health risks in children u5 accounting for malnutrition in Ghana and Pakistan. Draft background paper for the report Environmental health and child survival – epidemiology, economics, experiences. World Bank.

Larsen, B., 2007c: A cost-benefit analysis of environmental health interventions in urban Greater Dakar, Senegal. Paper for the Senegal Country Environmental Analysis, World Bank. Prepared from ECON/Roche Canada.

Larsen, B., 2005: Cost-benefit analysis of environmental protection in Colombia. Prepared for the Ministry of Environment, Housing and Land Development. Background paper for the Colombia Country Environmental Analysis: Colombia Mitigating Environmental Degradation to Foster Growth and Reduce Inequality. World Bank.

Larsen, B., 2004: Cost of Environmental Damage in Colombia- A Socio-Economic and Environmental Health Risk Assessment. Prepared for the Ministry of Environment, Housing and Land Development. Background paper for the Colombia Country Environmental Analysis: Colombia Mitigating Environmental Degradation to Foster Growth and Reduce Inequality. World Bank.

Larsen, B., 1997: Air pollution, health, and cleaner fuels: A cost-benefit analysis for Casablanca/Mohammedia. Background paper for Morocco Environment Review. World Bank.

Larsen, B., 1994: Air Pollution Projections and Costs and Benefits of Reductions in Iran. Background paper for Iran Environmental Strategy Study. World Bank.

Larsen, B. and Strukova, E., 2006: A cost-benefit analysis of improved water supply, sanitation and hygiene and indoor air pollution control in Peru. Prepared for the Peru Country Environmental Analysis: Environmental Sustainability a Key to Poverty Reduction in Peru. June 2007.

Larsen, B. and E. Strukova, 2005: Cost of environmental damage in Peru. Paper prepared for the Peru Country Environmental Analysis: Environmental Sustainability a Key to Poverty Reduction in Peru. June 2007. World Bank.

Lents, J., N. Davis, M. Osses, R., Nikkila, and M. Barth, 2004: Comparison of on-road vehicle profiles collected in seven cities worldwide. Presentation at Transport and Air Pollution, 13th International Scientific Symposium, 13-15 September 2004, NCAR Boulder Colorado, USA.

Li J, Guttikunda S, Carmichael G, Streets d, Chang Y-S and Fung V., 2004: Quantifying the human health benefits of curbing air pollution in Shanghai. Journal of Environmental Management. 70: p. 49-62.

Luo, Z. and Hulscher, W., 1999: Woodfuel Emissions. RWEDP, Bangkok. May 1999.

Mao X, Guo X, Chang Y and Peng Y., 2005: Improving air quality in large cities by substituting natural gas for coal in China: changing idea and incentive policy implications. Energy Policy. 33: p. 307-18.

Martins, MCH., F. Fatigati, T. Vespoli, L. Martins, L. Pereira, MA. Martins, P. Saldiva, and A. Braga, 2004: Influence of socioeconomic conditions on air pollution adverse health effects in elderly people: an analysis of six regions in Sao Paulo, Brazil. Journal of Epidemiology and Community Health, 58:41-46.

McCracken, J. and Smith, K., 1998: Emissions and Efficiency of Improved Woodburning Cookstoves in Highland Guatemala. Environment International. Vol. 24, No. 7.

Mehta S and Shahpar C., 2004: The Health Benefits of Interventions to Reduce Indoor Air Pollution from Solid Fuel Use: A Cost-Effectiveness Analysis. Energy for Sustainable Development. 8(3): p. 53-59.

Mestl, H.E.S., Aunan, K., Seip, H.M, Wang, S., Zhao, Y., and Zhang, D., 2006: Urban and rural exposure to indoor air pollution from domestic biomass and coal burning across China. The Total Science of the Environment (submitted).

Mrozek, J. and Taylor, L., 2002: What Determines the Value of Life? A Meta Analysis.
Journal of Policy Analysis and Management. Vol 21 (2): 253-270.

NAE/NRC, 2008: Energy Futures and Urban Air Pollution Challenges for China and the United States. National Academy of Engineering and National Research Council. Washington, D.C.: The National Academies Press.

Naeher LP, Leaderer BP, Smith KR., 2000: Particulate matter and carbon monoxide in highland Guatemala: indoor and outdoor levels from traditional and improved wood stoves and gas stoves. Indoor Air. Vol. 10, No. 3.

Netalieva I, Wesseler J and Heijman W., 2005: Health costs caused by oil extraction emissions and the benefits from abatement: the case of Kazakhstan. Energy Policy. 33: p. 1169-77.

Ostro, B. 2004: Outdoor air pollution – assessing the environmental burden of disease at national and local levels. WHO Environmental Burden of Disease Series, No. 5. World Health Organization, Geneve.

Pandey M and Nathwani J., 2003: Canada wide standard for particulate matter and ozone: cost-benefit analysis using a life quality index. Risk Analysis. 23 (1): p. 55-67.

Pinfold, J. and N. Horan, 1996: Measuring the effect of a hygiene behaviour intervention by indicators of behaviour and diarrhoeal Disease. Transactions of the Royal Society of Tropical Medicine and Hygiene, Vol 90.

Pope, CA III, R.T. Burnett, M.J. Thun, et al., 2002: Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. Journal of the American Medical Association, 287: 1132-1141.

Pye S and Watkiss P., 2005: Clean Air for Europe (CAFE) programme cost-benefit analysis. Baseline analysis 2000 to 2020. Study conducted by AEA Technology Environment for the European Commission, DG Environment.

Qian, Z., J. Zhang, L.R. Korn, F. Wei, and R.S. Chapman, 2004: Exposure-response relationships between lifetime exposure to residential coal smoke and respiratory symptoms and illnesses in Chinese children. Journal of Exposure Analysis and Environmental Epidemiology 14: S78–S84.

Rehfuess E, Mehta S and Prüss-Üstün A, 2006: Assessing household solid fuel use – multiple implications for the millennium development goals. Environmental Health Perspectives. 114(3): p. 373-8.

Reuters, 2007: China motor vehicle fleet hits 148 million. Reuters, April 10, 2007.

Saade, C., M. Bateman, and D. Bendahmane, 2001: The story of a successful public-private partnership in Central America: Handwashing for Diarrheal Disease Prevention. Published by BASICS II, EHP, Unicef, USAID, and World Bank.

Saatkamp, B., Masera, O., and Kammen, D., 2000: Energy and health transitions in development: Fuel use, stove technology, and morbidity in Jaracuaro, Mexico. Energy for Sustainable Development. Vol. IV, No 2. August 2000.

Salo, PM., Xia, J., Anderson Johnson, C., Li, Y., Kissling, GE., Avol, EL., Liu, C., and London, SJ., 2004: Respiratory symptoms in relation to residential coal burning and environmental tobacco smoke among early adolescents in Wuhan, China: a cross-sectional study. Environmental Health: A Global Access Science Source. 3:14.

Sinton, J.E., K.R. Smith, H. Hansheng, and L. Junzhuo. 1995: Indoor air pollution database for China." http://whqlibdoc.who.int/hq/1995/WHO_EHG_95.8.pdf.

Smith, K.R., Mehta, S., and Maeusezahl-Feuz, M., 2004: Indoor Air Pollution from Household Use of Solid Fuels. In Ezzati et al (ed): Comparative Quantification of Health Risks: Global and Regional Burden of Disease Attributable to Selected Major Risk Factors. Geneva. WHO.

Stevens, G., A. Wilson, and J. Hammitt, 2005: A benefit-cost analysis of retrofitting diesel vehicles with particulate filters in the Mexico City Metropolitan Area. Risk Analysis, 25(4): 883-899.

Song, Y., Y.H. Zhang, S.D. Xie, L.M. Zeng, M. Zheng, L.G. Salmon, M. Shao, and S. Slanina, 2006: Source apportionment of PM2.5 in Beijing by positive matrix factorization. Atmospheric Environment, 40:1526-1537.

UKDEFRA, 2006: Economic analysis to inform the air quality strategy review consultation: third report of the interdepartmental group on costs and benefits. Department for Environment, Food and Rural Affairs, April 2006.

UNEP, 2006: Opening the door to cleaner vehicles in developing and transition countries: the role of lower sulphur fuels. UNEP Partnership for Clean Fuels and Vehicles. Nairobi, Kenya. http://www.unep.org/pcfv/PDF/SulphurReport.pdf.

USEPA, 1999: The benefits and costs of the Clean Air Act: 1990 to 2010. Environmental Protection Agency, November 1999. http://www.epa.gov/oar/sect812.

USOBM, 2005: Validating Regulatory Analysis: 2005 report to congress on the costs and benefits of federal regulations and unfunded mandates on state, local and tribal entities. Office of Budget and Management, December 2005. http://www.whitehouse.gov/omb/inforeg/2005_cb/final_2005_cb_report.pdf

Venners, S.A., B. Wang, J. Ni, Y.G. Jin, J. Yang, Z. Fang, and X. Xu., 2001: Indoor air pollution and respiratory health in urban and rural China. International Journal of Occupational and Environmental Health 7:173–81.

Voorhees A, Araki S, Sakai R and Sato H., 2000: An ex post cost-benefit analysis of the nitrogen dioxide air pollution program in Tokyo. Journal of the Air and Waste Management Association. 50: p. 391-410.

WHO, 2002: World Health Report 2002. World Health Organization

WHO, 2007: Estimated deaths and DALYs attributable to selected environmental risk factors by WHO member state in 2002. World Health Organization. Geneve. www.who.int/quantifying_ehimpacts/countryprofilesebd.xls

Woodruff, T. J., J. Grillo, and K.C. Schoendorf., 1997: The relationship between selected causes of postneonatal infant mortality and particulate air pollution in the United States. Environmental Health Perspectives, 105(6):608-612.

World Bank, 2005: Islam Republic of Iran - Cost assessment of environmental degradation. Report No. 32043-IR. Washington D.C.

World Bank, 2007: World development indicators 2007. Washington D.C.

World Bank, 2008a: Environmental health and child survival – epidemiology, economics, experiences (forthcoming). Environment Department. Washington D.C.

World Bank, 2008b: Cost of pollution in China – economic estimates of physical damages (forthcoming). Washington D.C.

Xu, X., T. Niu, D.C. Christiani, S.T. Weiss, C. Chen, Y. Zhou, Z. Fang, W. Liang, and F. Zhang., 1996: Occupational and environmental risk factors for asthma in rural communities in China. International Journal of Occupational Environmental Health, 2(3):172–76.

Zhang, Y.H., X.L. Zhu, L.M. Zeng, et al., 2004. Source apportionment of fine-particle pollution in Beijing, In: Urbanization, Energy, and Air Pollution in China. National Academy of Engineering and National Research Council. Washington, D.C.: The National Academies Press. Pp. 139-153.

Zhang, W. et al, 2007: Source apportionment for urban PM10 and PM2.5 in the Beijing area. Chinese Science Bulletin, 52(5):608-615.

Zheng, M., et al, 2005: Seasonal trends in PM2.5 source contributions in Beijing, China. Atmospheric Environment, 39:3967-3976.