Valuing the health impacts from particulate air pollution in Tianjin

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Abstract

Although China has made dramatic economic progress in recent years, air pollution continues to be the most visible environmental problem and imposes significant health and economic costs on society. Using data on pollutant concentration and population for 2003, this paper estimates the economic costs of health related effects due to particulate air pollution in urban areas of Tianjin, China. Exposure-response functions are used to quantify the impact on human health. Value of a statistical life and benefit transfer are used to obtain the unit value of some health effects. Our results show significant health costs associated with air pollution in Tianjin. The total economic cost is estimated to be US$1.1 billion, about 3.7% of Tianjin's GDP in 2003. The findings underscore the importance of urban air pollution control. Finally, the policy implications for alternative energy options and climate policies are given.

Key words: particulate air pollution; PM_{10}; economic valuation; Tianjin

1. Introduction

China faces the challenges of both strengthening its economy and protecting its environment. Over the last decades, the intensified process of industrialization and urbanization, coupled with rapid population growth has resulted in severe environmental degradation. In particular, harmful pollutants such as sulfur dioxide (SO_2), nitrogen oxide (NO_2), ozone, total suspended particles (TSP) and particulate matter (PM) were emitted far exceeding the limits of national ambient air quality standards due to heavy reliance on coal as energy and rapidly growing motor vehicle fleet. According to the World Bank, China has 16 of the world's 20 most polluted cities [1]. More than 80% of the Chinese cities had SO_2 or NO_2 levels above maximum guideline levels set by WHO [2]. The current breakneck speed of industrialization is creating environmental problems of an unprecedented scale. Its implications are appalling since most of those cities are densely populated; the potential adverse health impacts due to air pollution can be considerable. Based on the World Bank estimates, pollution costs China in excess of US$54 billion a year in environmental degradation, loss of life and corresponding diseases [3]. Some 590,000 people a year in China will suffer premature deaths due to urban air pollution between 2001 and 2020, according to the "Vital Signs" report [2]. Recently, the Chinese authorities admitted that for all of their grand ambitions, there was only a little progress in pollution reduction and China had virtually failed to arrest environmental degradation in the past several years. Air pollution and its negative impacts on health and the environment are becoming a serious concern for both the public and the government in China.

Research in the past decades confirms that outdoor air pollution contributes to morbidity and mortality [4]. Although the biological mechanisms are not fully understood, state of art epidemiological studies have found consistent and coherent association between air pollution and various health outcomes. Observed effects include increased respiratory symptoms, reduced lung function, increased hospitalisations, chronic bronchitis, and mortality, especially respiratory and cardiovascular disease mortality [5-8]. Most of the epidemiological studies have focused on the effects of acute exposure, however, the effects of chronic exposure may be more important in terms of overall public health relevance [6]. Since the effects have been observed even at very low exposures, it remains unclear whether there is a threshold concentration below which no health effects are likely.

The associations between several health endpoints and pollutants have been quantified and the exposure-response relationships have been characterized. Among those pollutants, the role of suspended particulate matter has been particularly investigated regarding its effects on mortality and morbidity. Most of the recent studies report PM_{10} and $PM_{2.5}$ (particulate matter with an aerodynamic diameter of <10μg and <2.5μg, respectively) are the most responsible in life shortening effects although other pollutants may also associated with them [8-11]. The persistency of PM is mainly due to the fact that such fine particles can be inhaled deeply into the lungs where the clearance time of the deposited particles is much longer, thereby increasing the risk of adverse health effects [12].

The cost associated with adverse health outcomes from air pollution is largely borne by society. The external costs of air pollution have to be quantified before taking it into consideration in pollution control. Quantification of such health impacts would answer the cost-benefit question, such as 'what are the potential health gains by adopting pollution abatement policies or investing in the clean technology?' It would also provide an important message to policy makers about the severity of air pollution in terms of both health and economy. Health-related impacts from air pollution have been valued in monetary terms by many studies worldwide, particularly in the US and Europe [13-15]. There are also a couple of studies conducted for Shanghai, Liaoning and Shijiazhuang in China [16-18]. No results have been reported for Tianjin, however, although it is the third largest city of China with over 10 million people. As China attempts to move towards a more sustainable environment, the needs to measure, control and value air pollution are pressing.

This paper aims at assessing the cost of air pollution externalities, especially the adverse health impacts due to exposure to outdoor air pollutants in Tianjin municipality. While most of the epidemiological investigations are conducted for developed countries, it poses a big challenge and great concern about uncertainty whether those results could apply in China's context. However, in case there is no such study available in China, we believe that taking a value from elsewhere would be the best possible alternative in order to avoid a serious underestimation of the costs.

This paper proceeds as follows. Section 2 describes Tianjin municipality and its air quality, and estimates the air pollution level and population exposure. Section 3 introduces the methodology and data used in the study. Section 4 develops the monetary evaluation of the health impact of air pollution while Section 5 concludes the paper.

2. Tianjin and its air quality

Tianjin, China's third largest urban area and a major industrial center, plays an important role in the national economy. It is located in the northeast of the North China Plain and is the closest seaport to Beijing. As one of the municipalities under the direct administration of the Central Government of China, Tianjin is the economic center of China's Bohai rim and is being built into a modern port city and a major economic center in Northern China. Tianjin has a population in excess of 10 million, of which 59% are counted as urban, with most living in the city itself [19]. The municipality consists of six urban, three coastal, and six suburban districts and three counties. The urban districts in and around the city lie within its outer ring and occupy 254 km^2 while the municipality totals 11920 km^2.

The industrial output of Tianjin grew at 10% per year throughout the 1990s and its gross domestic product (GDP) reached US$29.5 billion in 2003. The economic boom has generated enormous demand for energy, the use of which in China has long been dominated by coal, accounting for more than 70% of the total energy consumption. The utilization of coal is the main emission source of greenhouse gases (GHG) such as carbon dioxide and other pollutants. Since the 1990s the energy use has gradually diversified from dominant traditional coal with electricity, oil and natural gas, as well as nuclear energy. The consumption of coal decreased from 76% of China's total energy use in 1990 to 66% in 2002. With the rapid increase in the numbers of motor vehicles in recent years, air pollution in large cities has gradually changed from coal combustion type to the mixed coal combustion/motor vehicle emission type [20]. Walsh [21] estimates the mobile vehicle emission is on average contributing to 45-60% of the NO_x emissions and about 85% of CO emissions in typical Chinese cities. In Tianjin the motor vehicles account for 40% of the NO_x emissions. In recent years the number of motor vehicles grew at an average rate of 15% per annum and most of the vehicles are domestically produced, and have a higher NO_x emission than the equivalent in developed countries. The emission discharge standard is also not effectively implemented since many of the cars exceeding the standard are still running on the streets. Furthermore, industrial expansion and population growth have concentrated serious environmental pollutants in densely populated areas. Air and water pollution, land degradation and noise become the major environmental concerns. Many of these issues end up being reflected in the health sector.

The air quality in Tianjin has worsened during the past decades due to its functioning as a heavy industrial center. The national air quality standards are exceeded most frequently for dust, SO_2 and CO. The major sources of air pollution are coal smoke and wind-blown dust, with the rapid growth in automobile pollution. But thanks to the environmental regulations and pollution control measures, air quality has improved slightly over the recent years. The national monitoring stations in Tianjin routinely measure ambient air quality. There are 12 monitoring sites located in the central districts. TSP, SO_2, NO_x and CO concentrations were the main measured pollutants before 2000. Since 2001, PM_{10} and NO_2 have been monitored instead of TSP and NO_x and reported on a daily basis.

Fig. 1 shows the trends of the average concentrations of major air pollutants in Tianjin from 1994 to 2003. The average concentration of TSP increased in the late 1990s but dropped noticeably since 2000, perhaps due to governmental regulations. SO_2 and CO pollution fluctuated over the years but in a downward trend. NO_x concentration has remained almost at the same level. The air quality monitoring data show that the annual average PM_{10} concentration in urban areas of Tianjin dropped from 167 $\mu g/m^3$ in 2001 to 133 $\mu g/m^3$ in 2003. The annual average SO_2 and NO_2 concentrations were 73 $\mu g/m^3$ and 51 $\mu g/m^3$, respectively in 2003 [22]. However, it is evident that current level of air pollutants is still quite high and all three indicators exceed the level of both China's air quality standards and WHO recommended guidelines (e.g. 50 $\mu g/m^3$ for SO_2 and 40 $\mu g/m^3$ for NO_2).

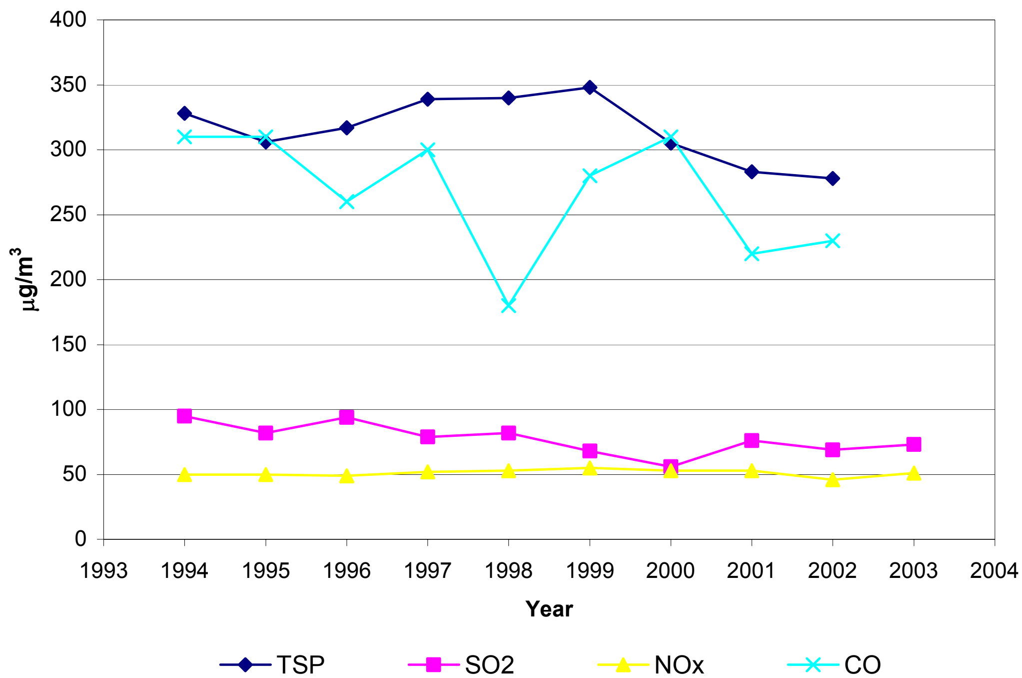

Fig.1. Annual average pollutants concentrations in urban area of Tianjin

Air quality is often poor, especially during the winter heating period. The monthly variations in Fig. 2 clearly show that PM_{10}, SO_2 and NO_2 concentrations all increased during winter in 2002. It is particularly obvious for SO_2, the concentration of which has aggravated significantly since October when the heating period starts and the concentration in winter is about four times higher than those during the rest of the year. PM_{10} concentration leaps up in March and April, which can be partly attributed to dust storms. In spring, the dust concentration is the highest and so are the total suspended and inhalable particles. Fang et al. [23] study the effect of dust storms on air pollution and concluded that PM_{10} concentration increases during dust storms. According to the Class I and II of national ambient air quality standards, marked with dashed lines in Fig. 2, PM_{10} violated the Class II of the standard (150 $\mu g/m^3$) for five months in 2002.

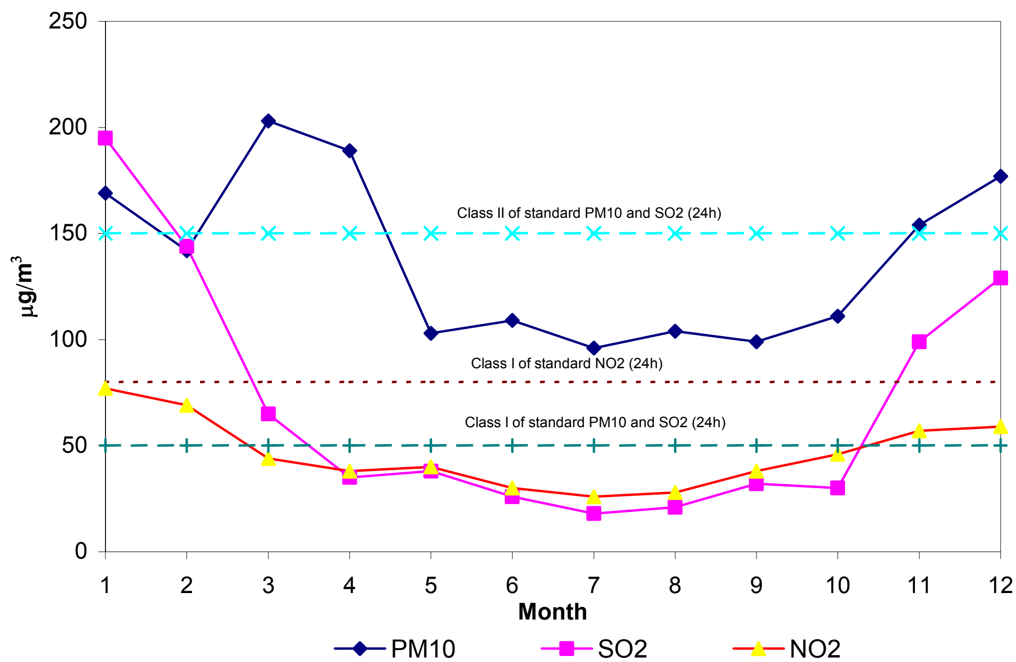

Fig. 2. Monthly variations of average pollutant concentrations in 2002

3. Methodology

A complex mixture of pollutants characterizes air pollution. Epidemiological studies use several indicators of exposure, such as TSP, PM_{10}, SO_2, NO_2, CO and ozone. However, these pollutants are correlated. It is hard to strictly allocate observed effects to a single pollutant. A pollutant-by-pollutant assessment would grossly overestimate the impact [24]. PM_{10} is regarded as an important and useful indicator for health risk of air pollution due to the ubiquitous nature of PM air pollution. Therefore, we selected PM_{10} as the main indicator for air pollution and used its impact on health as a proxy for the estimation of the economic cost of particulate air pollution.

Usually the concentrations of pollutants vary among monitoring sites and over the months, but due to limited access to data we could not develop a spatial model for emission and population exposure. Instead, this study adopts the annual average level of ambient concentration of PM_{10} of 133µg/m^3 in urban areas of Tianjin. Since the urban population concentrates in six urban districts, where the population density is particularly high and there is no information on population exposure from current research in Tianjin, in this study the whole urban population of 3.8 million is assumed to be exposed to particulate air pollution.

This paper adopts a three-step methodology to assess the costs of air pollution on health in Tianjin. Firstly, a set of health endpoints is established that is known to be associated with PM_{10} exposure. For each of them an exposure-response relationship is identified using the data published in the epidemiologic literature. The second step estimates the number of mortality and morbidity cases attributed to a given PM_{10} concentration level. Finally, we estimate the costs of increased cases of mortality and several endpoints of morbidity involving using benefit transfer and the value of a statistical life (VSL).

3.1 Estimation of health effects of air pollution

Published health impact assessment studies for particulate air pollution have mainly addressed mortality in people older than 30 years and excluding accidental causes, hospital admissions for respiratory and cardiovascular diseases, incidence of new cases of chronic and acute bronchitis and asthma attacks as well as restricted activity days (RADs) [9, 25-30]. Both long-term and short-term impacts on mortality can be estimated in terms of premature deaths. The impacts have been found in short-term studies, which relate day-to-day variations in air pollution and health, and long-term studies, which have followed cohorts of exposed individuals over time [7]. In China, due to the fact that inhabitants have long been exposed to poor air quality, the chronic respiratory incidence occurrence is much higher, while the acute effect tends to be lower [17]. In this context, we argue that a long-term mortality serves better in calculating the excess deaths than the short-term one. Therefore, we consider only here the long-term mortality. Table 1 lists the health endpoints examined in this study. Endpoints are ignored if the quantitative data were not available or if the costing was impossible (e.g. valuing the reduced lung function).

The association between particulate air pollution and health outcome frequency is usually described by an exposure-response function that gives the relative increase in adverse health for a given increment in air pollution. For each health outcome we rely on published epidemiological studies to derive the exposure-response function and the 95% confidence interval (CI). We prefer to derive the functions from the Chinese epidemiological studies if available. However, if no such study could be found on the health outcome that is considered, we use the coefficients from international peer reviewed studies. We argue that transferring the results from Western studies is justified because omitting the health endpoints would lead to the costs of air pollution being severely understated. When several studies provide information on exposure-response relationships, a meta-analysis of their results is conducted to derive a common estimate. Studies with high uncertainty are given less weight in deriving the final joint estimate. The function for long-term mortality is derived from two cohort studies Dockery et al. [9] and Pope et al. [25] in the US. The coefficients are obtained in part from Künzli et al. [24], a European study which conducted a meta-analysis from indigenous epidemiological studies, and Kan and Chen [16]. The data source and the chosen estimate and CI for 10 $\mu g/m^3$ of PM_{10} are shown in Table 1.

Table 1 Health endpoints and the exposure-response coefficients for particulate air pollution

Health endpoint (age group)	Exposure-response coefficient (95% CI)	Sources
Long-term mortality (adults ≥ 30 yr)	0.043 (0.026, 0.061)	Dockery et al. [6] Pope et al. [25]
Cardiovascular hospital admission	0.013 (0.007, 0.019)	after Künzli et al. [24]
Respiratory hospital admission	0.013 (0.001, 0.025)	after Künzli et al. [24]
Chronic bronchitis (adults ≥ 15yr)	0.045 (0.013, 0.077)	Ma and Hong [31] Jin et al. [32]
Bronchitis episodes (children <15 yr)	0.306 (0.135, 0.502)	after Künzli et al. [24]
Asthma attack (children <15 yr)	0.044 (0.027, 0.062)	after Künzli et al. [24]
Asthma attack (adults ≥ 15 yr)	0.039 (0.019, 0.059)	after Künzli et al. [24]
Outpatient visits-internal medicine	0.003 (0.002, 0.005)	Xu et al. [32]
Outpatient visits-pediatrics	0.004 (0.002, 0.006)	Xu et al. [32]
RADs (adults ≥ 20 yr)	0.094 (0.079, 0.109)	Ostro et al. [34]

Aunan and Pan [35] conduct a meta-analysis of the exposure-response functions for health effects of air pollution based on sixteen existing studies in China. They derived a 0.03% increase in all cause mortality per $\mu g/m^3$ PM_{10}, a 0.04% increase in cardiovascular deaths and a 0.06% increase in respiratory deaths. In addition, the study obtained a 0.07% and 0.12% increase in hospital admissions for cardiovascular and respiratory diseases, respectively, based on a study in Hong Kong. We selected the estimates from Künzli et al. [24] instead of the Hong Kong study because the former considered a couple of studies and derived the joint estimate as opposed to a single study. In general, the coefficients reported from the Chinese epidemiological studies tend to be somewhat lower than in the US and Europe, which may be due to a possible confounding with indoor air pollution or a misclassification of exposure [35].

The number of cases attributed to particulate air pollution in a given population is calculated based on three components: the exposure-response coefficients, the level of exposure and the observed incidence rates or prevalences of the health endpoints in the studied population. The attributable number of cases are calculated for each health endpoint, the algebra for the estimation being straightforward. The rationale is that the observed health outcome frequency rate is the rate in the population exposed to the corresponding concentrations. Using the exposure-response coefficient, one can calculate the frequency rate if the population were unexposed or exposed to a minimum level. The difference between this rate and the observed rate provides the attributable number of cases. This requires a clear definition of minimum exposure. The baseline frequency rate is usually derived from a log-linear or linear exposure-response function. For small exposure, the log-linear and linear would produce very similar results. However, extrapolating the function exponentially to a much higher exposure level than what the study is actually based on would seriously overestimate the effects. Therefore, in this study we follow a linear function. Since epidemiological studies on particulate air pollution have not been able to identify a concentration threshold for PM_{10} below which the adverse health impacts cease to occur, it may seem logical to use zero concentration as a reference. However, this is not the case. For example, Künzli et al. [24] used 7.5 μg/m^3 as a reference for PM_{10} as the natural background concentration. Quah and Boon [36] used both zero concentration and the minimum monthly PM_{10} of 24.7 μg/m^3 as a reference for Singapore. In a recent study by Kan and Chen [16], a natural background level of 73.2 μg/m^3 was used as a threshold for Shanghai. The estimated health impact can be calculated by the following formula:

$$\Delta E = \beta \cdot (C - C_0) \cdot F_0 \cdot Pop$$

where ΔE is the number of attributable cases to air pollution for each health outcome, C and C_0 refer to observed actual concentration and the threshold level respectively, F_0 is the baseline health outcome frequency rate under PM_{10} concentration of C_0, β is the exposure-response coefficient or relative risk which measures the associated increase per unit change in exposure level, and Pop is the population size exposed to C.

The baseline health outcome frequency rate should preferably be obtained from the actual rate of the population under study. In cases where such local data is not available, the health frequency data from other similar population may be used. In this study we derive our data mainly from the China Ministry of Health, the Tianjin Statistical Yearbook [19] and the Tianjin Municipal Bureau of Public Health. We also refer to Xu and Jin [17] for data for asthma attack and bronchitis episodes. The frequency rate for each health outcome we selected is shown in Table 2 in Section 4.

3.2 Valuation of the health effects to air pollution

3.2.1 Mortality

There are several existing methods to estimate the value of premature death. Earlier attempts to value mortality rely on human capital accounting measures, which are used extensively by calculating the discounted present value of net foregone earnings due to premature death. The method is criticized for ignoring the population which is economically inactive, such as infants and retired elderly although these people are the most vulnerable to air pollution related mortality [37]. Economists therefore have shifted to the more comprehensive "willingness to pay" (WTP) and "willingness to accept"(WTA) measures. These measures more completely capture the overall value of life by assessing the value that group of individuals place on reducing the risk of death or illness. Contingent valuation surveys, wage risk studies and consumer behavior studies are usually used to derive WTP. Value of a statistical life (VSL) is commonly used to express the value of mortality. As a result of different approaches used there is a widely varying empirical estimate of the VSL [38-39].

Ideally the VSL should be based on a WTP value in the study area. Since there is no such study for Tianjin and the method requires a large survey sample to ensure its reliability, a benefit transfer [40] is used to calculate the value of premature death. Benefit transfer approach involves the use of estimates of environmental loss of a project to estimate the economic value of environmental impact of a similar project on the assumption that the latter has the similar impact. In terms of mortality, this approach is to scale down the WTP by the ratio of per capita income of Tianjin to per capita income of the country where the value is adapted from. The ratio could be derived directly from income difference or from relative incomes by using purchase power parities (PPPs) as a conversion factor. This procedure also assumes that the income elasticity of WTP for improved health is 1.0 [41]. Some recent valuation studies have begun to address the issue of income and preferences in developing countries. Bowland and Beghin [42] derive a prediction function for developing countries which accounts for differences in income, estimating an income elasticity of WTP range of 1.52-2.27 for averted mortality. In contrast, Viscusi and Aldy [39] indicate an income elasticity of VSL from about 0.5 to 0.6 based on 60 studies from ten countries (mainly developed) in a meta-analysis. Quah and Boon [36] use a value of 0.32 for Singapore.

Only recently Chinese researchers begun to conduct local WTP studies for urban areas. For instance, Wang et al. [43] conduct a contingent valuation (CV) study in Chongqing, China and reported an average WTP for saving a statistical life of about US$34,750. Wang et al. [43] also calculate the marginal effect of income on WTP to be US$14,550 with an annual income increase of US$145.8, which implies an income elasticity of 1.8. The elasticity is quite high compared to Viscusi and Aldy [39] but lies well within the range suggested by Bowland and Beghin [42] for developing countries. According to this, we derived a WTP of US$85,833 for Tianjin after GDP per capita adjustment. Zhang [44] also uses the CV method to estimate for Beijing the WTP on mortality in 1999 and derived a unit value range from US$60,000 to US$200,000, which if transferred to Tianjin would be about US$70,055 to US$233,516 in 2003. This gives an average estimate of US$151,785. Since most of the VSL studies are conducted in the US and Europe, we need to compare the values in order to obtain a reasonable estimate for China. European Commission DG Environment [45] provides a best estimate of VSL of 1.0 million Euro for Europe-based mortality valuation for the year 2000, with a lower bound of 0.65 million and an upper bound 2.5 million. If the central estimate is transferred to China using PPP-based national output ratio between the European Union and China, then it is converted to the value for Tianjin based on the income differences between national average and Tianjin. We obtained a VSL of US$217,000 for Tianjin, which is much higher than the

Chinese estimates. We regarded this value as the upper bound of VSL and chose our first estimate US$85,833 as the lower bound. To generate a central estimate we computed the average of our high and low estimates. This produced a value of about US$151,410 as VSL for Tianjin.

3.2.2 Morbidity

As for valuation of the morbidity costs, we use both the WTP and the costs of illness (COI) approaches. COI measures the total cost of illness, including loss of human capital due to illness, the medical costs, such as hospital care, home health care, medicine, services of doctors and nurses, and other related costs. However, the indirect cost components, such as the opportunity costs of leisure, discomfort and inconvenience, as well as other intangible costs are neglected. Due to the very limited WTP literature on some of the health endpoints, we relied on the COI for estimating the unit values of hospital admissions, outpatient hospital visits and bronchitis episodes. Most of the cost data is from Tianjin itself. If the data for Tianjin is not available, we refer to the national average for large cities as an approximation. As a result, we derived a unit value of US$1302 and US$786 for cardiovascular and respiratory hospital admissions respectively; US$16 for bronchitis episodes and US$19.7 for outpatient visits.

For the remaining endpoints, such as chronic bronchitis, asthma attacks, RADs, the WTP values are estimated from other studies and transferred to China and Tianjin taking into account the income differences, using an income elasticity of 1.0. For example, the unit value of chronic bronchitis is derived from the World Bank study [46] and applied to Tianjin using an income elasticity of 1.0. This produced a unit value of US$10370 per case. The values for asthma attacks per case and RADs per day are converted from the European Commission report, which yields a unit value of US$11.5 and US$18, respectively.

4. Results

This section presents the results of the study, including the attributed cases to each health endpoint, the unit value of each health impact and the total economic cost of those impacts due to particulate air pollution in Tianjin. In estimating the attributed costs we used the baseline frequency rate, concentration level of PM_{10} and population exposure. The annual average PM_{10} concentration of 133 μg/m^3 in 2003 was taken for the estimation. The threshold is assumed to be 50 μg/m^3, which is the national primary ambient standard for PM_{10} in China. Table 3 shows the frequency rate under the actual exposure and the attributed number of cases due to air pollution given this threshold level. The central estimates as well as the lower and upper estimates from the confidence interval are given.

Table 2. Attributable number of cases

Health endpoint (age group)	Frequency rate per person per year	Attributable number of cases		
		Central	Lower	Upper
Long-term mortality (adults ≥ 30 yr)	0.0094	5965	4025	7623
Cardiovascular hospital admission	0.0115	4234	2387	5922
Respiratory hospital admission	0.0050	1841	156	3248
Chronic bronchitis (adults ≥ 15yr)	0.0124	11226	4021	16097
Bronchitis episodes (children <15 yr)	0.0310	10038	7393	11283
Asthma attack (children <15 yr)	0.0824	9948	6808	12635
Asthma attack (adults ≥ 15 yr)	0.0831	67640	37678	90924
Outpatient visits-internal medicine	0.7025	64523	43367	105825
Outpatient visits-pediatrics	0.1355	16465	8367	24308
RADs (adults ≥ 20 yr) [a]	3.5	4521709	4085912	4900535

[a] per person-day per year

In total, particulate air pollution caused 5,965 attributed premature deaths in urban areas of Tianjin in 2003, which accounts for about 25% of annual adult deaths in the population under study. For lower and upper estimates, the percentages are 16% and 31%, respectively. It also results in 4,234 new cases of cardiovascular hospital admissions and 1,841 cases of respiratory admissions. In addition, it accounted for 11,226 new cases of chronic bronchitis, 10,038 bronchitis episodes, 77,588 asthma attacks, 80,988 outpatient visits and 4.5 million restricted activity days.

Table 3 summarises the selected unit values for mortality and morbidity effects with the corresponding types of estimates and presents the total economic cost of air pollution. Based on the unit value for each health endpoint and the attributed cases, we computed the total health damage cost associated with particulate air pollution in Tianjin. In the central estimate, the total economic cost is US$1.1 billion, or 3.7% of GDP in Tianjin municipality in 2003. Among them, the mortality costs is predominant, accounting for about 80% of the total economic costs of air pollution. The morbidity cost is about US$208 million, accounting for 20% of the total costs, in which chronic bronchitis contributes the greatest, followed by restricted activity days and cardiovascular and respiratory hospital admissions. The lower and upper estimates are US$730 million and US$1423 million accounting for 2.5% and 4.8% of Tianjin's GDP, respectively.

Compared to the estimates for other Chinese cities, the value is not particularly high. Kan and Chen [16] assess the economic cost of air pollution for Shanghai, which equivalent to 1.03% of GDP in 2001. Xu and Jin [17] estimate the cost for Fushun, Liaoning and find that it accounts for 0.75%-1.95% of GDP in the year 2000. Peng et al. [18] produce a value of 4.3% of GDP for the cost of air pollution in Shijiazhuang in 2000.

Table 3. Unit values of health endpoints and the total economic cost of air pollution

Health endpoint	Unit value (US$)	Approach	Total cost (million US$)		
			Central	Lower	Upper
Premature death	151410	WTP	903.18	609.49	1154.18
Cardiovascular hospital admission	1302	COI	5.51	3.11	7.71
Respiratory hospital admission	786.0	COI	1.45	0.12	2.55
Chronic bronchitis	10370	WTP	116.42	41.69	166.92
Bronchitis episodes	16.0	COI	0.16	0.12	0.18
Asthma attacks	11.5	WTP	0.89	0.51	1.19
Outpatient visits-internal medicine	19.7	COI	1.27	0.85	2.08
Outpatient visits-pediatrics	19.7	COI	0.32	0.16	0.48
RADs	18.0	WTP	81.39	73.55	88.21
Total			1110.59	729.61	1423.51

As mentioned earlier, the estimation of the cost was conducted based on a threshold of 50 $\mu g/m^3$. Since the value of the threshold is critical in determination of the total economic cost of air pollution, we performed a sensitivity analysis with respect to various threshold levels. Table 4 shows the total economic costs if the thresholds are 0 $\mu g/m^3$ or 75 $\mu g/m^3$, and compares with the present estimate. The total cost increases by US$417 million, which is about 1.5% of Tianjin's GDP in 2003 if using zero threshold. In contrast, the total cost decreases by 0.8% of GDP with a higher threshold of 75 μg/m^3. In addition, our results suggest a per capita cost of air pollution of US$294 for Tianjin in 2003.

Table 4. Sensitivity analysis of total costs by various thresholds of PM_{10}

Threshold	Mortality cost (million $)	Morbidity cost (million $)	Total cost (million $)	Cost as portion of GDP (%)	Cost per capita (US$)
Assumed value (50 μg/m^3)	903.2	207.4	1110.6	3.7	293.8
No threshold	1249.3	277.9	1527.3	5.2	404.0
Natural background (75 μg/m^3)	685.4	161.1	846.5	2.9	223.9

The uncertainty of this study stems from several factors. Firstly, some of the exposure-response functions are transferred from the US or European context to Tianjin in which they may not be applicable in a stricter sense, given the differences in the exposure level, social and economic characteristics of the population under investigation, as well as the physical and chemical composition of particulate particles across countries or regions. More precise estimation would require a large number of domestic epidemiological studies, including cohort studies, which may take China years and many efforts to develop. The second limitation is the selection of PM_{10} as the main indicator of air pollution, which may underestimate to some extent the adverse health impacts since other particles and pollutants also contribute to them. The particulate air pollution emphasizes the importance of outdoor air pollution, the indoor pollution from smoking, stove heating and cooking are largely neglected. After all, people spend more time indoors than outdoors, which is especially true in winter months when heating is needed. In Tianjin, indoor air pollution generating from the use of raw coal and biomass for cooking and heating poses great health risks to some of the population. Thirdly, the unit values of each health outcome are rough estimates, which is associated with some kind of uncertainty especially with respect to the VSL and benefit transfer. Transferring the results from one area to another entails uncertainty itself due to differences between regions in many aspects such as income, age distribution, culture and health status. In spite of the above limitations, this study provides useful information regarding the health impacts of particulate air pollution as a first attempt to value air pollution in Tianjin.

5. Conclusions and policy implications

Using the PM_{10} concentration level, population at risk of air pollution, exposure-response function, as well as unit values of various health effects, this study provides some rough estimates of the health impacts of particulate air pollution in Tianjin. Our results suggest that the cost imposed on society is substantial in both absolute and relative terms. The total economic cost of particulate air pollution amounts to US$ 1.1 billion, accounting for 3.7% of Tianjin's GDP for the year 2003. It implies that the reduction of ambient concentrations of PM_{10} would yield substantial health benefits that are equal to a very significant percentage of the GDP. The results obtained deliver a clear message to relevant policy makers about the importance of controlling air pollution and the potential gain in the health sector. The potential benefits may exceed the present estimate since the impact of some other pollutants is not considered and the estimate is regarded as a conservative one. It also underscores the importance of implementation of policies and strategies for mutual development of economy and environment. Under business as usual scenario the environment will aggravate further and one day to a level in which some adverse impact would be irreversible and enormous economic, as well as welfare loss would incur.

The possible policy initiatives for reducing air pollution include energy conservation and emission control measures. As Tianjin still benefits substantially from rapid economic growth that is based on cheap coal and gas, the energy consumption is most likely to increase. However, a more efficient energy structure and adopting advanced technology would help reduce pollutants and GHG emission considerably. One option is to reduce the coal consumption by gradually replacing coal with natural gas, which largely depends on the availability of gas and the transmission infrastructure. Another option is to use lower sulphur coal to replace coal, which would greatly reduce sulphur pollution. Peng et al. [18] suggest replacing the small boilers with larger and more efficient facilities that reduce energy consumption and emission in heating sector. According to the Tianjin Blue-Sky project implementation scheme in 2003, Tianjin has effectively controlled pollution related to the coal burning by replacing coal in the smaller boilers with cleaner energy sources or removing them altogether [47]. Besides physical measures to reduce emission, economists have suggested to impose a tax or charge on emission discharge. Without internalising the health costs through polluter pays principle, natural gas and central heating systems will not be competitive with coal-based small boilers. China has employed the Pollution Levy System but it has not been effectively implemented because the levies are well below the marginal cost of control and also below the operating costs of the pollution control equipment. Facilities thus have an economic incentive to pay the levy rather than operate the equipment especially since levies are treated as an operating expense for tax purposes [48].

The second policy implication is the long-term development and use of motor vehicles with tighter pollution standards and higher energy efficiency. Transport has become a major source of air pollution in recent years with the rapid rise of motor vehicle numbers in China. Walsh [49] suggests an 8% annual growth of fleet till 2020 under the medium GDP growth assumption. The corresponding demand for fuel would be high as well. Due to rapid increase of the vehicle population, NO_x, CO and ozone problems will be increasing. In view of the very rapid growth in the vehicle fleet forecast, China's environment could face severe strains with significant public health consequences unless vehicle technology is substantially upgraded and fuel quality improved. According to Walsh [49], an approach to dealing with rising energy consumption, in addition to improving vehicles and fuel, is to substantially increase the use of alternative fuels such as Dimethyl-ether (DME) and Fischer-Tropsch diesel, producing them from China's coal reserves. In addition, He et al. [50] project road transport related oil consumption and CO_2 emission in China and found that under no control scenario the oil demand in China's transport will grow at an annual average rate of 6.1% and reach 363 million ton in 2030 and the corresponding CO_2 emissions will increase five-fold by 2030.

The last implication for policy makers is to address the linkages between the local air pollutants and GHGs. Most of the traditional air pollutants examined in this study and GHGs have common sources. As a developing and non-Annex I country, under the Kyoto Protocol, China is not bound to any GHG abatement targets during the first control period (from 2008 to 2012). However, the country has a right to sell GHG emissions obtained to industrialized countries under a trading mechanism of this Protocol, the so-called Clean Development Mechanism. Therefore, by mitigating GHGs emitted from intensive carbon source, Tianjin may create a 'win-win' solution. Cao [53] investigates the options for mitigating GHG emissions in Guiyang, China and conducted a cost-ancillary benefit analysis. This paper concludes that it is possible to achieve simultaneously the abatement of GHG emission as well as local air pollutants such as SO_2 and particulate matters in

China by adopting various climate friendly technology options related to coal consumption. Thus emissions control strategies that simultaneously address air pollutants and GHGs may lead to a more efficient use of the resources on all scales.

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