Economic Analysis of Selected Environmental Issues in China

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Abstract

This thesis investigates several selected environmental issues in China from an economic perspective. It consists of four self-contained papers. Chapters 2-4 address the issues related to water shortage, while Chapter 5 focuses on the cost of air pollution.

Chapter 2 analyses the potential application of desalination in China from an economic perspective. Concerned with water shortage in China, the study aims to assess the potential of desalination as a viable alternate water source through the analysis of the costs of desalination, the water demand and supply situation, as well as water pricing practices in China. The study shows that there is a significant decline in the costs of desalination for two main processes over time. The average unit cost of US\$0.6/m^3 for desalting brackish water and US\$1.0/m^3 for seawater, are suggested to be feasible for China. The future trends and challenges associated with water shortages and water pricing are discussed, leading to conclusions and recommendations regarding the role of desalination as a feasible source of water for the future.

Chapter 3 extends the cost analysis of Chapter 2 from two to five desalination processes and evaluates the cost of water transport. The unit costs of desalinated water are evaluated, followed by multivariable regressions to analyse the main influencing factors to the costs. The results show that the unit costs for all the processes have fallen considerably over the years. The regressions show that the total installed capacity, the year, the raw water quality, and the location of the plant all play a role in determining the unit cost of desalination. Transport costs are estimated to range from a few cents per m^3 to over a dollar. A 100m vertical lift is about as costly as a 100km horizontal transport (0.05-0.06\$/m^3). Therefore, transport makes desalinated water prohibitively expensive in highlands and continental interiors, but not elsewhere.

Chapter 4 focuses on the econometric analyses of domestic, industrial and agricultural water uses in China using province-level panel data. The study shows that the regional disparity in the level and pattern of water uses is considerable. Economically developed or more industrialised areas at the coast consume less water than the agriculture dominated provinces in the west and far south of China. The results suggest that both economic and climatic variables have significant effects on water demands. For the domestic sector, income is the dominant factor influencing the magnitude of water use and shows an income elasticity of 0.42. We find that richer provinces have a higher income elasticity than do poorer provinces.

Chapter 5 values the health impacts from air pollution in Tianjin. 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 concentrations and population, the study estimates the economic costs of health-related effects due to particulate air pollution in urban areas of Tianjin. The results suggests the total economic cost is about US\$1.1 billion, or 3.7% of Tianjin's GDP in 2003. The findings underscore the importance of urban air pollution control.

Key words: water shortage, air pollution, economic analysis, desalination, water use, MSF, RO, PM_{10}, external cost, China.

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Chapter 1

General introduction

1.1 Environmental problems in China

China has seen tremendous economic growth in recent years, becoming a dominant economic force in the world in the 21st century. Rapid industrialisation, urbanisation and social changes have raised the standard of living for millions of Chinese people. But the success comes at great environmental cost. China has followed a pattern similar to many developed and developing countries where the process of industrialisation is strongly linked to deteriorating environmental quality. This pattern usually only turns around when a country has improved its standard of living (Dinda, 2004).

China's rapid economic growth over the last two decades has reduced the country's natural resources and caused severe environmental degradation. Vaclav Smil's "China's Environmental Crisis: An Inquiry into the Limits of National Development" (1993), Lester Brown's "Who Will Feed China"(1995) and Elizabeth Economy's recent book "The Rivers Run Black: the Environmental Challenge to China's Future" (2004) are typical for the literature that spread the notion of "China's environmental crisis". Plausible estimates of environmental costs vary from 3-15% of China's GDP (Varley, 2005).

Among all the environmental problems in China, water shortage and air pollution have been the most prominent and remain to be tackled. Most rivers in North China run dry and water managers struggle to meet demands with limited and sometimes declining water resources. Of the 640 major cities in China, more than 300 face water shortages, with 100 facing severe scarcities (NEPA, 1997). Water shortages are often accompanied by water pollution. According to a recent report published by the State Environmental Protection Administration (SEPA), the seven major rivers and 25 out of 27 major lakes in China are polluted, some seriously. The most polluted river in the country is Haihe River in the north, followed by Liaohe River, Huaihe River, Yellow River, Yangtze

River and Pearl River (People's Daily, 2005). Water pollution throughout the country has caused significant health-related problems, including rising rate of cancer and respiratory diseases. The impact of China's dual problem of water scarcity and water pollution exacts a costly toll on productivity. Water shortages in cities cause a loss of an estimated 120 billion yuan (US$14 billion) in lost industrial output each year (SPC, 1995). The impact of water pollution on human health has been valued at approximately 33 billion yuan per year (US$3.9 billion), which is almost certainly an underestimate (World Bank, 1997). Future economic development continues to be jeopardized by water shortages.

Besides water problems, air pollution is becoming the most visible environmental problem associated with industrial growth. According to the World Bank, China has 16 of the world's 20 most polluted cities (Economist, 2004). Coal, which supplies more than three-quarters of China's electricity, is the major source of air pollution. Sulphur dioxide and soot caused by coal combustion are two major air pollutants, resulting in the formation of acid rain, which now falls on about 30% of China's total land area (SEPA, 1998). Industrial boilers and furnaces consume almost half of China's coal and are the largest single point sources of urban air pollution. In recent years rapid growing motor vehicle fleet has also become one of the major sources of urban air pollution. Air pollution, especially suspended particulate matter and sulphur dioxide are causing enormous respiratory and pulmonary diseases in China.

Motivated by the severe water problems that China is facing, this thesis investigates several important aspects of water shortage from an economic perspective. Firstly, water shortage is addressed from the standpoint of supply management in which advanced technology, i.e. desalination, is explored in terms of its development and costs to provide additional water. The costs of seawater and brackish water desalination are evaluated and compared to the water prices, and the feasibility and potential applications of desalination in China are assessed. Secondly, water shortage is addressed by considering demand management measures for which the water demands in major water-using sectors are examined and analysed and the underlying factors that affect water demands are investigated. In addition, this thesis devotes one chapter to the assessment of the external cost of air pollution by taking the case of Tianjin. The economic analyses in this thesis contribute to improving the understanding of economic, technical and institutional aspects of water and air problems in China and to the comprehension of the interactions between environment and economy.

1.2 Water conditions of China

China's rapid economic growth, industrialisation and urbanisation, coupled with inadequate infrastructure investment and management capacity, have all contributed to the widespread problems of water scarcity throughout the country. With a total amount of 2800 billion m^3 of annually renewed freshwater, equivalent to a per capita water resource of 2220 m^3, less than one third of the world average, China faces some of the more extreme water shortages in the world. By 2050 this volume is estimated to decrease to around 1700 m^3 (Liu and Chen, 2001). The absolute value, however, does not entirely reflect the water conditions in China because water is not spatially and temporally evenly distributed. About 80% of water resources are located in the Yangtze River and its southern part where the population is only about 50% of the total (Liu and Chen, 2001). North China is especially water-poor, with only about 750 m^3 per capita, which is one-fifth of the per capita water in southern China and just 10% of the world average (Figure 1.1). The distribution of groundwater is also skewed: average groundwater in the south is more than four times greater than in the north. Dramatic shifts in monthly and annual precipitation cause floods and droughts, which threaten the economy and people's lives.

As surface water is reduced, groundwater has been increasingly extracted to meet the demands. In some places, groundwater has been exploited in excess, leading to a continuous decline in the water level, triggering environmental processes, such as subsidence, ground cracks and seawater intrusion. For instance, in Tianjin, as a result of over-pumping, the groundwater table fell in the 1980s with a rate of 0.5m per year on average, and in some places up to 1m per year. Groundwater levels in some areas have fallen by as much as 40m since the late 1970s, with some spots pumped down to the bedrock (TBWR, 2000). Over-extraction of groundwater has become a serious problem in many other cities in the north e.g. Shijiazhuang, Taiyuan, and Xi'an as well as in a number of coastal cities including Dalian, Qingdao, and Yantai. Although there is no comprehensive monitoring data on groundwater, studies suggest that the quality, not just the quantity, is severely threatened in many regions. Groundwater pollution occurs in nearly half of all urban areas in China. Of the total national groundwater resources, only 63% are usable as drinking water without treatment, 17% can be used for drinking after appropriate treatment, 12% are unsuitable for drinking but can be used as industrial and agricultural water sources, and 8% can be used as industrial water only after special treatment (ITA, 2005).

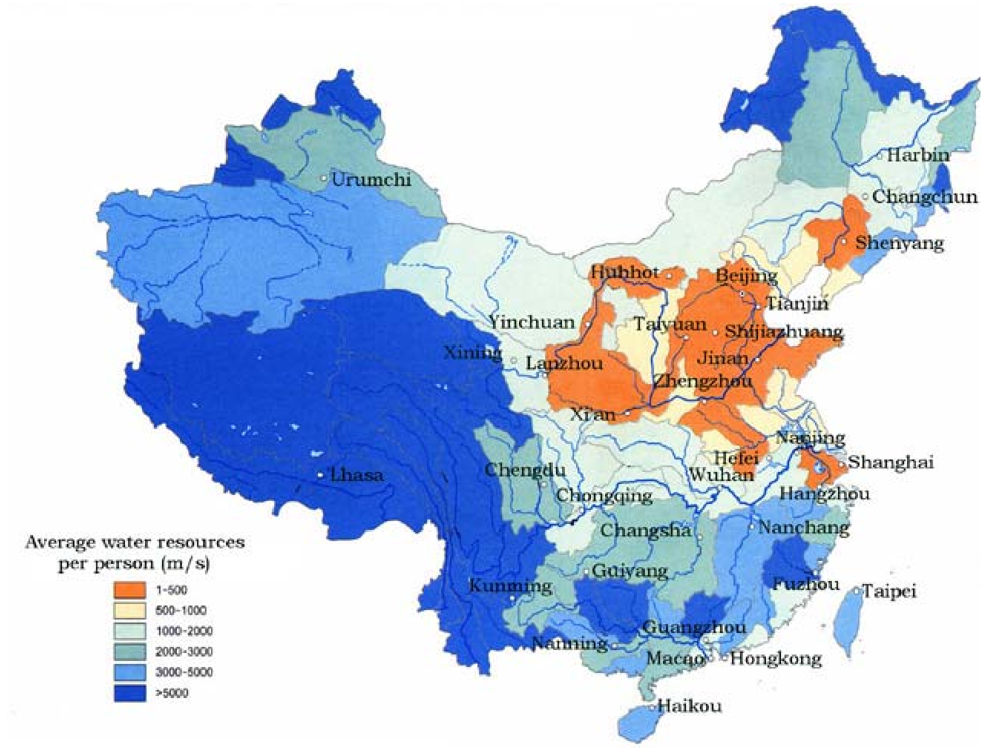


Figure 1.1 Distribution map of average water resources per person (based on the population in 1997) (Liu and Chen, 2001)

Each year large amounts of pollutants are discharged into China's water bodies from municipal, industrial and agricultural sources. Statistics available from the Ministry of Water Resources show that more than 70% of China's rivers and lakes have been polluted to varying extents. Except for some inland rivers and large reservoirs, water pollution has worsened in recent years, with the pollution near industrial cities and towns being particularly severe. The main sources of water pollution include point sources, such as industrial discharge and municipal sewage, and non-point sources consisting of agricultural runoff and scattered township or village enterprises. The pollution not only contaminates the river body but also entails a potential threat to human health. Water pollution is closely interlinked with water shortage. On one hand, pollution reduces the amount of freshwater while on the other, lower water flows coupled with higher wastewater discharge impair the river's self-purifying function, which otherwise would be achieved if the discharge is less than 20% of the water flow. Some of the major threats stem from inadequate treatment of both industrial and municipal wastewater. The total amount of wastewater discharged in 2002 was 63.1 billion m^3. Industrial wastewater accounted for 61.5% while domestic made up 38.5% (China Statistical

Yearbook, 2003). The amount of municipal wastewater treated in 2002 was 13.5 billion m^3, with a treatment rate of about 40%, which is far from adequate given China's serious water pollution. In counties, towns and extensive rural areas, wastewater treatment rates were significantly lower. A large amount of wastewater is still being discharged directly into surface water bodies without treatment. The actual wastewater treatment rate in China may be less than 20% (ITA, 2005).

Water shortages affect China's economic and social development, particularly in the north. A growing population, rapidly developing economic and social system, accelerated urbanisation and improvement in standard of living imply a greater gap between water supply and demand as the country develops. The traditional measures to address water shortage are mainly supply-oriented and aim at fostering the development and exploitation of new water sources and expansion of the network infrastructure. Under the current dry situation in North China, expansion of the capacity in water storage and groundwater pumping would not help much in obtaining additional water. The South-North water transfer scheme is a recent project of this kind, which attempts to transport water from the Yangtze River to rivers in the north. However, this project is costly and entails ecological impacts (Wang and Ma, 1999).

Desalination, as a fast-growing technology, is promising in providing more water by converting sea- or brackish water into freshwater. Desalination has expanded rapidly in recent decades, with the total installed capacity growing from 8000 m^3/day in 1970 to about 32 million m^3/day by 2002 (Wangnick, 2002). It has allowed socio-economic development to continue in many arid, semi-arid and other water-short areas. The application has been very noticeable in parts of the Middle East, North Africa, the Arabian Gulf and some islands where traditional water supply cannot meet the needs. Desalted water has become an alternative to traditional water supply and has increasingly been explored by many other regions (Wangnick, 2002). The development of desalination is driven by the increasing stress on the water sector, which cannot satisfy the ever-growing water demand generated by population and economic growths and more water-consuming lifestyles. It is also driven by the reduction of costs of desalination due to technological improvements and improved management and experience.

China's population and economy are concentrated in the coastal zone, which makes desalination a viable alternative source of water, as many coastal cities face water shortage. The potential application of desalination depends on the economical as well as technical feasibility in China. Technical application seems not an important problem if provided with sufficient training and technical know-how. The remaining concern is whether desalination is affordable for the government and the Chinese people at present as well as in the future. In order to answer this question, an adequate knowledge on the costs of various desalination technologies is crucial. The current and future water shortages and water pricing practices in various sectors are also important. In this thesis, a study on the implications of desalination for water resources in China is conducted from an economic perspective. This study is then extended to a cost analysis on desalination and water transport, in which desalination costs of all the main technologies are assessed and the cost of water transport is evaluated.

In recent years, besides the supply-oriented measures to tackle water scarcity, water policies have increasingly addressed demand management, which means development of management programs to reduce water demand and conserve water. Demand-driven measures include adoption of water saving technologies and appliances, awareness-raising and economic instruments, such as taxes. Concerning the increasing costs of developing new water supply and dealing with the existing inefficiency in the system, an initiative to adopt conservation and water use efficiency measures and a move towards demand management seems urgently needed in China. In some water-scarce places like Tianjin, water saving technologies and appliances, as well as recycling of water have been implemented. At a larger scale, however, demand management measures are still rarely adopted in China. Agriculture uses the largest share of China's water, accounting for about 65% of the total, followed by industrial use (22%) and domestic use (12%)(CWRB, 2003). Low water efficiencies lead to a large amount of water squandered, which is particularly obvious in irrigation. For demand control measures being effectively implemented, the patterns and levels of water uses in various sectors across the regions need to be comprehended. The factors that influence water demand of each sector need to be understood.

1.3 Air pollution and health effects

Intensified industrialisation and urbanisation have caused severe degradation in air quality. In particular, harmful pollutants, such as sulphur dioxide (SO_2), nitrogen oxide (NO_2), ozone, total suspended particles (TSP) and particulate matter (PM) have been 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. Ambient concentrations of TSP and SO_2 are among the world's highest. SEPA tests in more than 300 cities in China indicate that air quality in almost two-thirds fail to achieve WHO standards for acceptable level of TSP. Some major cities have SO_2 well above the WHO standard of 60 $\mu g/m^3$, which means that about 600 million people are exposed to the level above the standard (Varley, 2005). Among the air pollutants, particulate matter PM_{10} (less than 10 microns in aerodynamic diameter) is the most dangerous because such fine particles can be deeply inhaled into the lungs where they may be deposited, resulting in adverse health effects.

Poor ambient air quality prevails in most cities in China. Considering that more than 450 million of China's 1.3 billion people are now living in urban areas, the poor air quality could have considerable adverse health effects. China's failure to meet the residential ambient air quality standards exposes a large population to health risks, such as chronic bronchitis, pulmonary heart diseases and lung cancer. 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 (Worldwatch, 2005). Although China has attempted to arrest air pollution by enforcing environment-friendly programs, the problem remains serious and the air quality has not noticeably improved. 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.

Air pollution is an externality. Quantification of such costs is significant in analysing the benefit of adopting pollution abatement policies and investing in clean technologies. Health-related impacts from air pollution have been valued in monetary terms by many studies worldwide, particularly in the US and Europe (e.g. US EPA, 1999; Monzon and Guerrero, 2004; and Danielis and Chiabai, 1998). There are only very few studies carried out in China. As China attempts to move towards a more sustainable environment, it is urgent to measure, control and value air pollution. In this thesis, a valuation study on the cost of air pollution is conducted for urban areas of a heavy industrialised city.

1.4 Outline of the thesis

This thesis contains four papers, which are presented in Chapters 2-5. The articles are written in a way that each can be read independently although some of them are closely connected. Chapters 2-4 address problems related to water shortage while Chapter 5 focuses on the external cost of air pollution.

To solve or eliminate water shortage problems, seawater desalination draws more and more attention as an alternative water supply source. The costs of water produced by desalination have dropped considerably over the years as a result of reductions in the price of equipment and power consumption, and advances in system design and operating experiences. In Chapter 2 the implications of desalination for water resources in China are analysed from an economic perspective in order to answer the question: "Is it economically and practically feasible to apply desalination in China?" Since desalination plants have not been constructed on a reasonable scale in China, the costs for two main desalination processes, multistage flash distillation (MSF) and reverse osmosis (RO) are assessed, using the data available for desalination plants worldwide. Based on the investment costs and estimated operation and maintenance costs, an economic appraisal for the costs of desalination for these two processes has been conducted. There are a few studies that have conducted a cost comparison analysis, but these either compare a limited number of plants with a single process, compare different technologies in a single plant, or compare plants on a regional basis. This study, however, extends previous studies to a global scale by reviewing and analysing the average costs of various desalination plants in countries all over the world and illustrates the trends of the costs in order to make a suggestion for the potential application in China. This study also evaluates China's current water supply and demand situations, as well as future projections, and discusses the water pricing practices. The results of the study provide an overview of the projected costs of desalination, current and future water shortage and potential applications of desalination in China. It also serves as a basis for developing governmental plans, strategies and policies for future applications of desalination.

In Chapter 3, we extend the analysis presented in Chapter 2 to evaluating the costs of desalination and water transport. This study expands the cost analysis of desalination to all major desalting technologies, including multiple effect evaporation (ME), vapour compression (VC) and electrodialysis (ED), and assesses the cost of water transport over a distance. The study offers a comprehensive cost analysis of desalination for the first time. This is interesting because many regions of the world that are facing freshwater scarcity are looking for technically and economically feasible alternatives. Such cost information on desalination, as compared with the local cost of water supply, could provide a sense of potential application of desalination in a region. The study defines the main economic parameters used in estimation of desalination costs and calculates the unit costs of desalted water for five main processes based on simplified assumptions. It then uses multiple variable regressions to estimate the trends of unit costs over time and to analyse the significant factors that influence the cost of desalination. Moreover, in this study a literature survey on the costs of water transport is conducted in order to estimate the total cost of desalination and the transport of desalinated water to the regions with water shortage. Chapter 3 provides insight into the development of desalination for water managers and policy makers, which can be used to assess the potential of desalination plants in a certain region.

In Chapter 4, econometric analyses are conducted for water demands in domestic, industrial and agricultural sectors. This chapter aims to shed light on the estimation of water demands in various sectors in China and to enhance the understanding of the factors that influence the demand. Water consumption has increased substantially in 1990s due to population growth, rapid urbanisation and overall expansion in economic activities. However, little empirical research has been done on water demand estimation of these major sectors in China. The main reason is the lack of time-series and cross-sectional survey data. There are several urban household water use surveys conducted in the municipalities Beijing and Tianjin. The surveys are normally conducted over a short period of one to two years and for a relatively small sample size. The survey results are largely presented in qualitative terms regarding the current household characteristics, housing, water using appliances and amenities, water consumption levels, as well as water use behaviour and perception (Zhang and Brown, 2005). This study attempts to bridge the gap by providing a comprehensive analysis on water uses and also for the first time using panel data for China. Climatic variables are also for the first time included in such an analysis. The province level panel data from 1997 to 2000 is used firstly to examine the regional disparity in the level and pattern of water uses and secondly to conduct econometric analyses of water uses in different sectors. The models for domestic, industrial and agricultural water uses are developed separately, taking into consideration households' income, characteristics, water prices, water availability and weather variables. The models are estimated using feasible generalised least square (FGLS) because of heteroskedasticity and autocorrelation. The results of this study are of direct relevance to water resources planning and policy making. The estimates of elasticities can be used in water demand forecast or in cost benefit analysis of future water supply projects.

In Chapter 5, the economic cost of air pollution in Tianjin is evaluated. 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. We chose Tianjin as a case study because it is a typical industry-driven city in China. Its air quality has worsened during the past decades, as Tianjin is a centre of heavy industry. The health cost of air pollution resembles the cost for hundreds of other similar industrial cities of a different scale. In this study a three-step methodology to assess the costs of air pollution, particularly particulate matter, on health in Tianjin is adopted. Firstly, a set of health endpoints is established that is known to be associated with PM_{10} exposure and for each of them an exposure response relationship is identified using data published in 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 using benefit transfer and the value of a statistical life (VSL). The data on air pollutants in urban areas of Tianjin and population structures for 2003 are used. The results of the study not only deliver a clear message to relevant policy makers about the importance of controlling air pollution and the potential gain in the health sector but is also of direct relevance to cost benefit analysis in emission reduction and pollution control measures.

Chapter 6 contains the summary of the results and conclusions. The policy implications and recommendations for future research are given.

Chapter 2

Implications of desalination for water resources in China: an economic perspective[1]

2.1 Introduction

China is a country with great variations in the spatial and temporal distribution of its water resources. There is more than sufficient water in the south but there is a water deficiency in the north. North China has suffered from water shortages for the past few decades, and due to the population growth and economic development, this region has now reached the level of severe water scarcity. Poor water condition has been a factor restricting the socio-economic development and causing environmental deterioration. Traditional water supply cannot help to provide more water to meet growing demands. The South-North Water Transfer Scheme attempts to ease water problems by transporting water from the Yangtze River in the south to rivers in the north, which is only a choice due to a lack of alternatives. The project is by far the largest infrastructure construction of China in terms of investment and complexity (Chinawater, 2002).

However, improvements in desalination technology may pave the way to more accessible water. China's population and economy are concentrated in the coastal zone, which makes desalination a good alternative source of water as many coastal cities face water shortage. This study analyses the implications of desalination to water resources in China from an economic perspective in order to answer the question: "Is it economically and practically feasible to apply desalination in China?" Since desalination plants have not been constructed on a reasonable scale in China, the costs for two main desalination processes, MSF and RO are analysed, using data available for desalination plants all over the world. The research also evaluates the water situation and future projections of China. The results of the study provide an overview of the projected costs of

[1] Chapter 2 is published as Zhou, Y. and Tol, R.S.J. (2004), Implications of Desalination for Water Resources in China: An Economic Perspective, *Desalination,* 164(3), 225-240.

desalination, current and future water shortage in China, and potential applications of desalination in China. It also serves as a basis for developing governmental plans, strategies, and policies for future applications of desalination.

2.2 Current state of desalination

Desalination of seawater and brackish water has grown rapidly in recent decades. This has allowed socio-economic development to continue in many arid, semi-arid and other water-short areas. The application has been very noticeable in parts of the Middle East, North Africa, the Arabian Gulf and some islands where traditional water supply cannot meet the needs. Desalted water has become an alternative to traditional water supply and has increasingly been explored by many regions. The installed capacity of desalination plants has expanded rapidly worldwide, from 8000 m^3/day (in 1970) to about 32 million m^3/day (by 2001). Non-seawater desalination plants contributed with 13.3 million m^3/d, whereas the capacity of the seawater desalination plants reached 19.1 million m^3/d (Wangnick, 2002). The development is driven by the increasing stress of the water sector, which cannot satisfy the ever-growing demands for water generated by population growth, economic growth and more water-consuming lifestyles. It is also driven by the reduction of costs of desalination due to technological improvements and improved management and experience.

Various distillation and membrane technologies are available for seawater and brackish water desalination, including multiple effect distillation (MED), multistage flash distillation (MSF), reverse osmosis (RO) and electrodialysis (ED). The first two are based on distillation process whilst the latter two use membrane technology. The most important and popular processes are MSF and RO, which account for 84% of the whole capacity of the world (Gleick, 2000). Most of above-mentioned processes can apply to desalt seawater, while RO and ED are often used for brackish water desalting. The selection of different technology essentially depends on the purposes of desalination, economics, the physical conditions of the plant site, raw water and product water qualities, and local technical know-how and capacity.

2.3 Desalination costs

One of the most important factors determining desalination decisions is economics: costs and benefits. However, it is not easy to analyse and compare the costs of different desalination plants, because the costs strongly depend on the capacity and type of plants, the region, the quality of raw and product water, the period and assumptions about capital and labour costs. Fortunately, there is indeed a trend that the cost of desalination has been declining over years. To get a general understanding of the costs and their trends, it is important to conduct a cost comparison of existing desalination plants.

There are a few studies that have conducted a cost comparison analysis, but these studies either compare a limited number of plants with a single process, compare different technologies in a single plant, or compare plants on a regional basis (Ebrahim and Abdel-Jaward, 1999; Tian and Wang, 2001; Ashraf and Pablo, 1999; Ali El-Saie et al., 2001 and Park et al., 1997). Park has conducted a comprehensive cost comparison using 1990 unit cost for analysing the potential of desalination in Korea, but used plant data of only the period from 1982 to 1991 (Park et al., 1997). The Desalination Economic Evaluation Program (DEEP) developed by the International Atomic Energy Agency has been applied to some studies for economic evaluation and screening analyses of various desalination and energy source options in the world (Gowin and T. Konishi, 1999).

In China there are very few desalination plants of a reasonable scale in use at present; therefore, it is not feasible to make a cost analysis based on them. This study reviews and analyses the average costs of various desalination plants in countries all over the world based on simple assumptions, and then illustrates the trends of decline in order to make a suggestion to the potential application in China. A huge number of desalination plants are considered and classified into several groups based on desalination technologies. The main data of desalting plants in this study are obtained from 2002 IDA Worldwide Desalting Plants Inventory Report No.17 (Wangnick, 2002). Since MSF and RO are to date the most often used processes, account for most of the capacity, plants using these two processes are selected and their costs are analysed and compared. For the purpose of this study and simplicity, the plants are only classified by process, disregarding the location, the quality of source and product water, and other specific conditions.

The major costs elements for desalination plants are capital costs and annual operation and maintenance costs (O&M). Capital costs can be divided into direct and indirect costs. The direct costs include the costs of purchase of equipment, land, construction charges and pre-treatment of water. The indirect costs mainly refer to the interest, insurance, construction overheads, project management and contingency costs. Annual operation costs are those expenses incurred during actual operation, such as labour, energy, chemicals, consumables and spares. Calculations of unit product costs depend on the process, the capacity, site characteristics and design feature.

For this study, all the plants using the MSF and RO processes in IDA Report No. 17 are included, which contain about 3000 data points from 1950 up to now. The data set includes country, location, total capacity, units, process, equipment, water quality, user, contract year and investment costs. The investment costs should firstly be amortised, which can be obtained by multiplying these costs by an amortization factor. The formula is as follows:

$$A = P \times i \times (1+i)^{n-1} \Big/ \left[(1+i)^{n} - 1 \right] \tag{2.1}$$

where A is amortised annual capital cost, P is the value of investment in the original year, i is the annual discount rate, and n is the economic plant life. In this study, a discount rate of 8% and a plant life of 25 years are assumed for amortization for all cases as these figures are usually used in this sector in both China and other countries (Wangnick, 2002 and MOC, 1993). Due to the lack of data for operating costs, 60% of total cost is assumed to be operating costs for all the cases (Wangnick, 2002). For the purpose of comparison, all costs must be evaluated based on the same year level. As all the costs have been converted to US dollar, the base year 1995 is selected and all costs are converted according to the United States Consumer Price Index. The costs data include investment costs, amortised capital costs, O&M costs, total unit cost, conversion rate and 1995 unit costs (see appendix).

2.3.1 Cost comparison of the MSF process

The MSF process accounts for the second largest installed desalting capacity for the world. The major consumers for MSF are in Saudi Arabia, United Arab Emirates and Kuwait. Figure 2.1 shows the yearly distribution of the unit costs of desalting plants in the world. The unit costs decline over time, from about 9 $/m^3$ in 1960 to about 0.9 $/m^3$

in 2000. Since the MSF process is mostly applied for seawater desalination plants, the costs reflect the value of desalting seawater. The trend indicates that the desalting costs of seawater are expected to decrease further in the future. Based on the exponential projection presented in Figure 2.1, the average cost will go down to about 0.3 $\$/m^3$ in 2025. As the costs have fallen by a factor of 10 in 40 year's time, a further cost decrease by a factor of 3 in 25 years is entirely feasible. This value, however, is associated with great uncertainty because of the crudeness of the underlying data described in section 2. As China is a country which lacks experiences in seawater desalination, the current estimated cost would be a bit higher than the average world level, perhaps about 1.0 $\$/m^3$ would be an appropriate cost of the MSF process in China at the moment, and lower in the future.

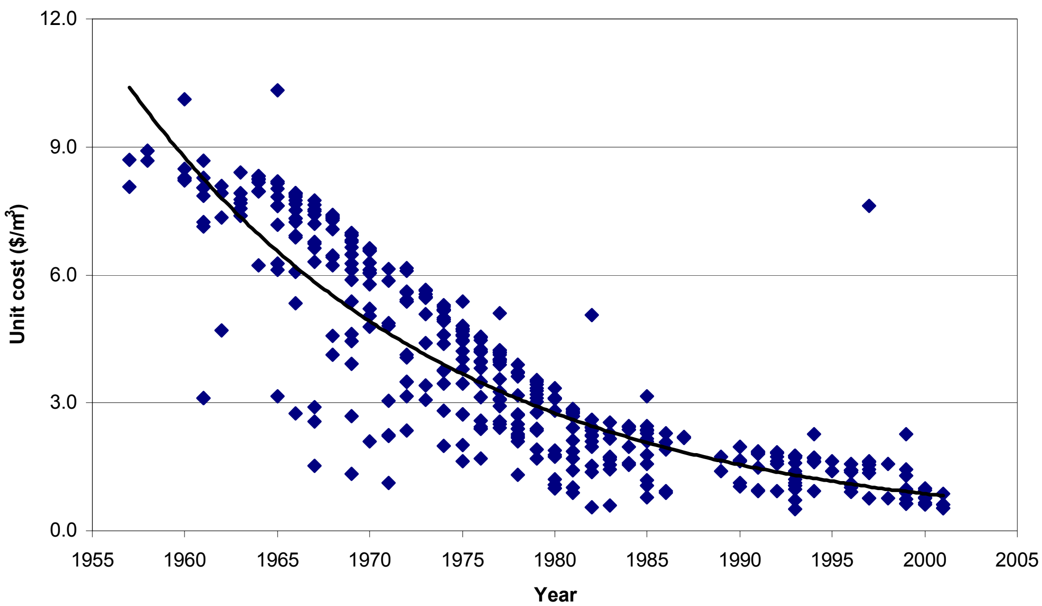


Figure 2.1 Yearly distribution of the unit costs by MSF process

Figure 2.1 illustrates the trend of unit costs with plant capacity. As shown, there is a decline of cost with the increase of plant capacity due to economies of scale. However, the trend is not pronounced as the points are distributed dispersedly along the trend line for plants with a capacity less than 50,000 m^3/d. This may result from many other influencing factors besides capacity, such as the quality of raw and product water, the costs of labour and energy.

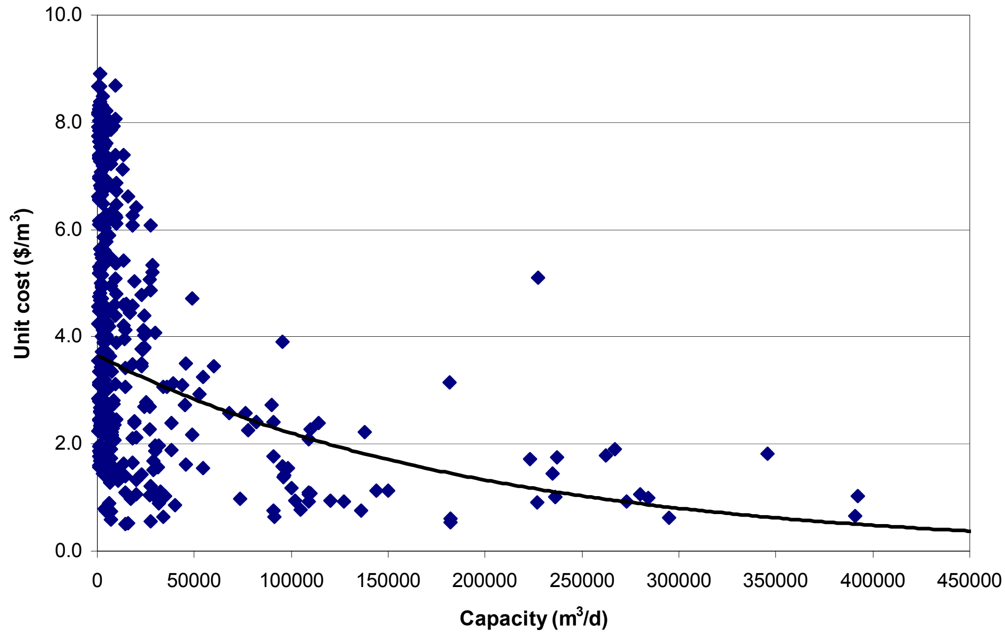


Figure 2.2 Distribution of the unit costs with plant capacity by MSF process.

2.3.2 Cost comparison of the RO process

The RO process has become more popular during the last decades due to a significant achievement in improving technology. At present, RO has the largest share of the total installed capacity in the world. The operating cost of RO plants has been reduced thanks to two developments: 1) lower-cost, higher-flux, higher salt-rejecting membranes that can operate efficiently at lower pressures and 2) the use of pressure recovery devices (Gleick, 2000). Figure 2.3 shows the distribution of the unit costs with the total installed capacity by the RO process. As shown, the unit costs have declined with the cumulative installed capacity as a result of the technological development and gained experiences. Compared to the costs of the MSF process, the costs of RO have been much lower. According to the trend, the unit cost will continue to decrease, as more and more desalting plants are built in the future.

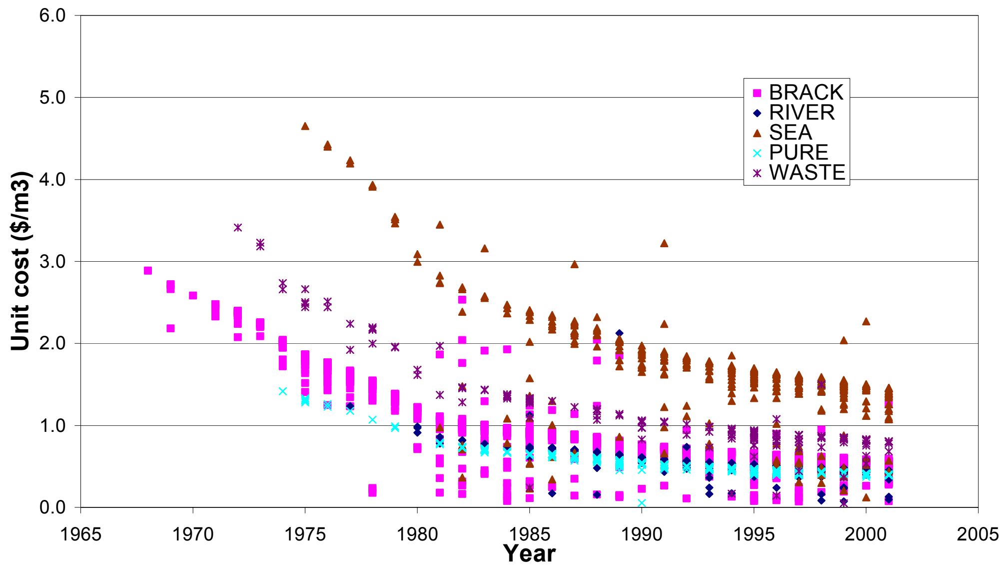


Figure 2.3 Distribution of the unit costs with total installed capacity by RO process.

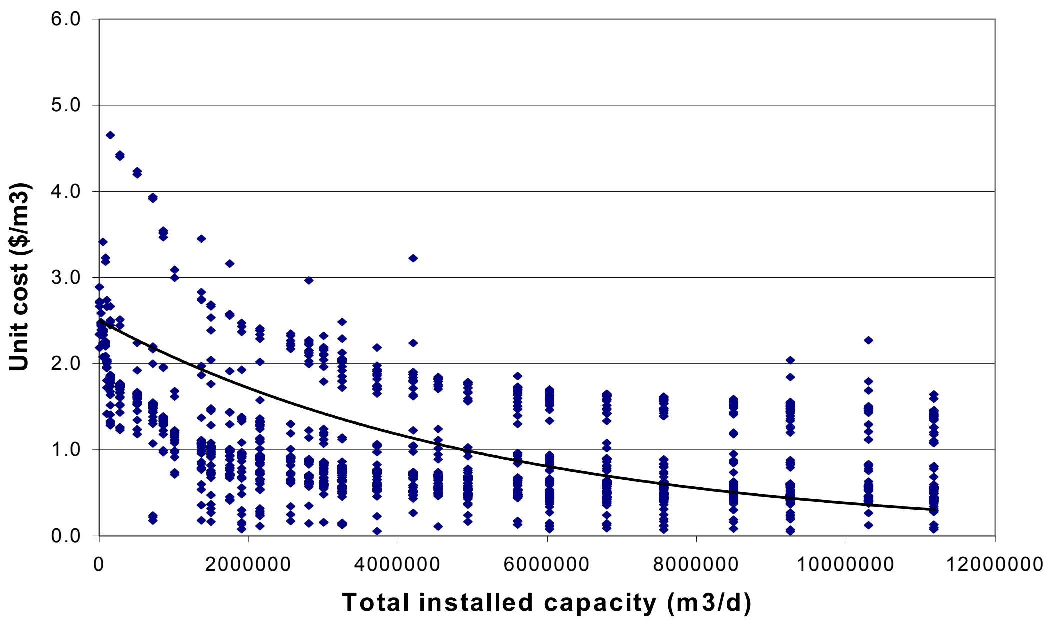


Figure 2.4 Yearly distribution of the unit costs by raw water quality for RO process.

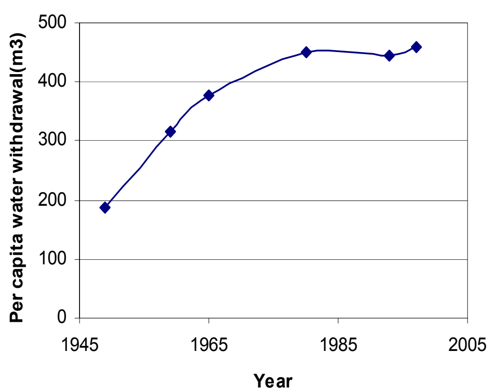
Unit costs vary with the raw water quality, location, capacity and so on. Figure 2.4 shows the variation of unit costs with different raw water qualities, namely brackish-, sea-, river-, waste- and pure water. The unit costs have declined considerably over time. The average cost of desalting by RO process goes down to about 0.7 $/m^3$ in 2000. RO is used more often to desalt brackish water, river and pure water although it is also increasingly applied to seawater. From our calculation, we can see that the average costs for desalting brackish water are lower than for seawater and wastewater desalting, but higher than for river and pure water. The costs of seawater desalting have been going down, however, in 2000 they are still above 1.0 $\$/m^3$. For brackish water, the average cost has decreased to about 0.5 $\$/m^3$ today. Great interest and efforts have been put in seawater RO research in the last decade, which makes its costs, especially on a larger scale, reduced considerably. Recent tenders' lower costs of large SWRO plants indicate that RO has great potential to become the most economical process for seawater desalination. Most of the RO plants have a smaller capacity than MSF plants in general. Given the fact that the desalination plants with small or medium scales will be the most suitable at the beginning of desalination in China, the RO process is likely to be the first choice. Today a cost of 0.6 $\$/m^3$ for desalting brackish and wastewater and a cost of 1.0 $\$/m^3$ for desalting seawater by RO would be valid in China.

2.4 Implications of desalination for water resources in China

2.4.1 Water resources in China

China has a total amount of 2800 km^3 of water resources. According to the 1997 population statistics, the average volume of water resources per capita is only 2220 m^3. Based on this index, the country ranks as the 121st place among all other nations in the world (Liu and Chen, 2001). By the next 50 years, the volume is estimated to go down to around 1700 m^3, which reaches the threshold of water stress (Liu and Chen, 2001). The absolute value, however, does not reflect the reality of water resources because water is not evenly distributed in both spatial and temporal terms in China. For example, about 80% of the total volume of water is located in the Yangtze River and its southern part of China, where the population accounts for 53.6% of the total and the area is only 35.2% of the whole country (Liu and Chen, 2001). The per capita water resources in the south are much greater than in the north. With regard to the temporal variation, 70% of the total precipitation is concentrated mostly during four months of the year (Liu and He, 2001).

The water withdrawal has increased dramatically in recent decades, from 443.7 km^3 in 1980 to 556.6 km^3 in 1997 (Liu and Chen, 2001). The water withdrawal per capita is illustrated in Figure 2.5, increasing to 458 m^3/year in 1997. The utilisation ratio (the percentage of water withdrawal out of water resources) also rose from 16.1% in 1980 to 19.9% in 1997 (Liu and Chen, 2001). In North China, water resources are over exploited and the utilisation rate reaches 50% or more, including Huang River, Huai River and Hai River (Figure 2.6). According to the international criteria, more than 40% withdrawal can be regarded as a threshold of severe water scarcity. The overdraft has resulted in discontinuous flow, declined groundwater table, and degradation of the ecological system. The future growing demands and requirements will further deteriorate the water situation, thus constrain the socio-economic development of the region.



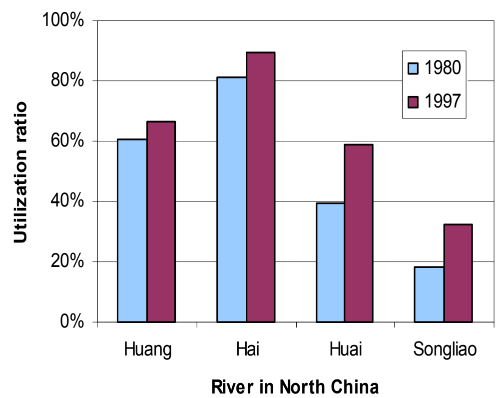


Figure 2.5 Water withdrawal per capita (m^3/yr) Figure 2.6 Utilisation rate of water in North China

2.4.2 Future water demands

To assess the future potential of water resources, it is not feasible to merely rely on the volume data. It is equally important to forecast future water use including changes human activities. A report from the Chinese Academy of Engineering has recently been released, projecting water resources and water demands for the next 50 years (Liu and Chen, 2001). The projection includes industrial demand, domestic demand and agricultural demand, taking into consideration the population growth and socio-economic development (urbanisation, industrialisation, change of industrial structure, etc). Three scenarios are applied for the projection (Table 2.1), namely high economic growth (HG), moderate economic growth (MG) and low economic growth (LG). Different assumptions regarding economic development, government policy, and increase of irrigation area are given for these three scenarios. The population projection is shown in Figure 2.7, and GDP projections for the three scenarios are illustrated in Figure 2.8. As shown, in 2050 the population in China will grow to 1.6 billion, and GDP will increase to about 100 trillion Chinese yuan ($12 trillion) under moderate growth.

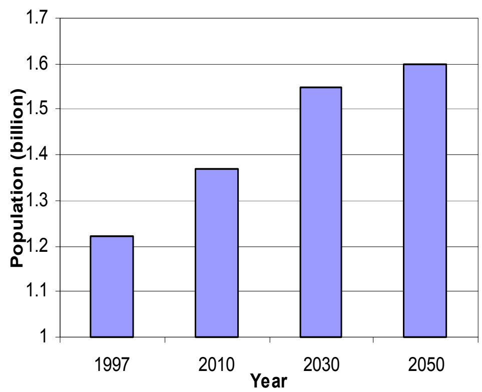


Figure 2.7 Population projections in China

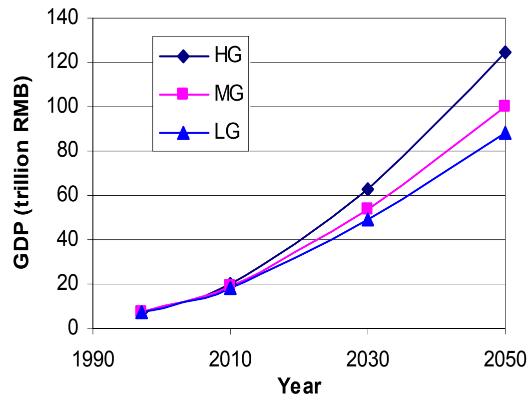


Figure 2.8 GDP projections under three scenarios

As shown in Table 2.1, the water demands in 2050 are projected to reach 800 km^3 under high economic growth and about 700 km^3 under low development. The increase in water demand in comparison with current level is estimated to be between 130 to 230 km^3, which is a huge amount of water. The Chinese government has built many dams, reservoirs and other infrastructure to exploit water in the past, which makes the expansion of storage capacity difficult because many infrastructures have been in place and developing more hydraulic projects tends to be costly. This implies that the future water resources are increasingly difficult to meet the growing needs. Under the moderate growth scenario the water demands in 2050 will be 730 km^3, with per capita demand of 457 m^3/yr. The structure of future water demand will change as follows: agriculture will use comparatively less water while the industrial and residential sector will increase water demands. It is estimated that in 2050 the ratio of using water in agriculture-industry-domestic sectors will be 57:27:16 as compared to the 1997 ratio of about 71:20:9 (Liu and Chen, 2001).

Table 2.1 Water demands projection in the next 50 years (km^3/yr) (Liu and Chen, 2001)

Year	High Growth			Moderate Growth			Low Growth		
	Total	North	South	Total	North	South	Total	North	South
1997	571.4	273.9	297.5	571.4	273.9	297.5	571.4	273.9	297.5
2010	659.1	310.6	348.5	642.4	303.7	338.7	630.5	297.4	333.1
2030	757.3	346.9	410.4	711.9	330	381.9	688	319.1	368.9
2050	806.3	366.2	440.1	731.9	337.1	394.8	702.7	323.1	379.6

There are several other projections regarding population and GDP growth of China. We use two of them for a sensitivity study of future water demands. Within this study, two of the IPCC SERS scenarios (Nakicenovic and Swart, 2000) were compared with China's projection. In the context of this study, only population varied among different scenarios whilst all the other variables remained constant. Table 2.2 shows that under scenario A_1, in which population grows slowly, water demands will fall by 7.5% (55 km^3) in 2050 in comparison to China's projection. With the higher population projection of scenario A_2, water demands will increase by 13.1% (96 km^3). From this sensitivity study and the sensitivity analysis on economic growth (GDP) reported above, we see that the uncertainty about future water demands of China is high.

Table 2.2 Sensitivity analysis of water demands under different population scenario

Year	China scenario		IPCC scenario A1			IPCC scenario A2		
	Population billion	Demands km^3	Population billion	Demands km^3	Deviation %	Population billion	Demands km^3	Deviation %
2010	1.37	642.4	1.35	638.6	0.6	1.45	655.0	2.0
2030	1.55	711.9	1.41	685.0	3.8	1.76	749.9	5.3
2050	1.60	731.9	1.32	676.8	7.5	2.09	827.7	13.1

Table 2.3 Water shortage analysis for Huang, Huai and Hai River basin (km^3/yr) (Liu and Chen, 2001)

River	Year	Water Availability*	High Growth		Moderate Growth		Low Growth	
			Demands	Shortage	Demands	Shortage	Demands	Shortage
Huang	2030	44.3	57.1	12.8	53.5	9.2	52.3	8
	2050	44.8	60.5	15.7	54.5	9.7	53	8.2
Huai	2030	73.5	85.3	11.8	81.5	8	79.9	6.4
	2050	76.4	89.7	13.3	83.9	7.5	81.6	5.2
Hai&Luan	2030	40.6	56.1	15.5	53.9	13.3	52.7	12.1
	2050	41.8	58.7	16.9	55.6	13.8	53.7	11.9
Total	2030	158.4	198.6	40.2	188.9	30.5	184.9	26.5
	2050	163	208.9	45.9	194	31	188.2	25.2

* including current water transfer capacity

Apparently water resources cannot grow symmetrically with increasing demands for water consumption; hence water shortage will become strikingly severe, especially in North China where the utilisation ratio of water is already extremely high at present. Table 2.3 shows water demand and supply conditions of three main river basins in North China, namely Huang River, Huai River and Hai River. The data of water availability include the potential utilisation of surface and groundwater, potential wastewater reuse as well as current water transfer capacity of 15 km^3 per year. However, it does not include the future water transfer from other rivers. Table 2.3 shows that water shortage will be 46 km^3/yr under the HG scenario, 31 km^3/yr under the MG scenario and 25 km^3/yr under the LG scenario in 2050. If population growth is higher than expected, shortages would increase even further.

The water supply under the current scheme will not be able to meet the future demands, thus inter-basin water transfers for a large scale have been considered and approved. Water transfers from Yangtze River to North China are collectively known as the South-North Water Transfer Scheme. This scheme has mixed impacts on natural environment: it on one hand can provide a stable source of water for receiving basins and on the other hand has negative environmental impacts. The general layout of the scheme has been worked out as three water transfer projects, namely Western Route Project (WRP), Middle Route Project (MRP), and Eastern Route Project (ERP), which will divert water from upper, middle, and lower reaches of Yangtze River respectively, to meet the developing requirements of Northwest and North China (MWR, 1995)(see Figure 2.9). The preliminary estimate of the total capital investment is about 500 billion yuan (about \$60 billion). The total transfer capacity of the West Route, Middle Route and East Route Project is estimated to be about 44.8 km^3/yr by 2050 (MWR, 2002).

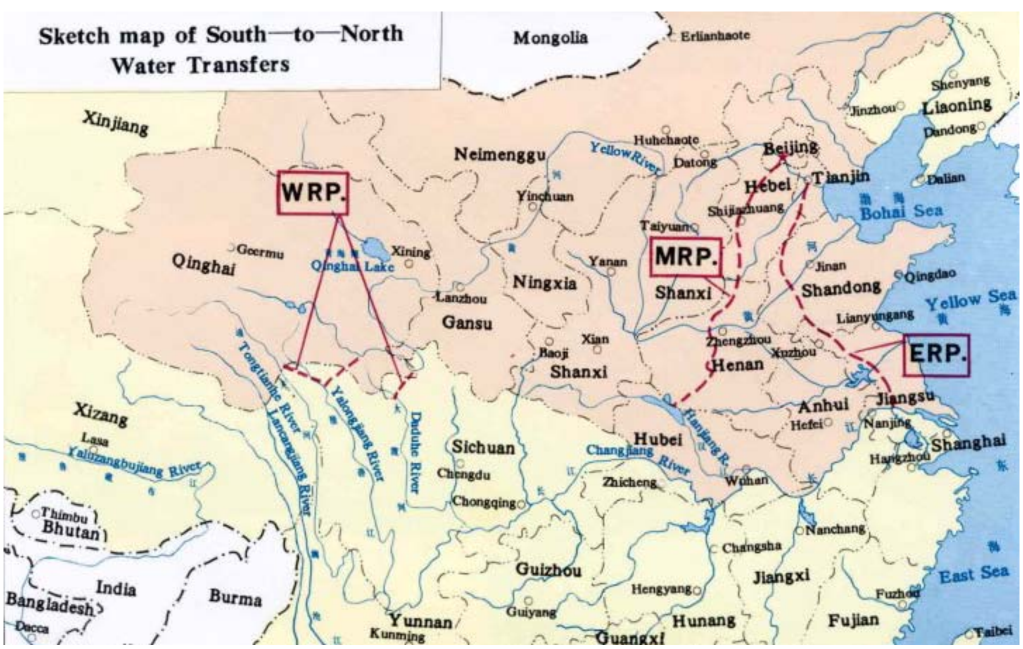


Figure 2.9 Map of the South-North water transfers

The ERP will divert water from the lower reach of Yangtze River north to supply water for the eastern Huang-Huai-Hai Plain with the termination in Tianjin City by raising water in stages through Beijing-Hangzhou Grand Canal. The Ministry of Water Resources estimates that the first round expansion of the route can complete in 3 years, and the second phase could be completed by 2010, with another round of expansion considered for after 2010. The MRP will divert water from Danjiangkou Reservoir on the Haijiang, a tributary of Yangtze River, to Beijing City through Canals to be built along Funiu and Taihang Mountains. The advantages of this project lie mainly in good quality of the water to be diverted, greater availability of water supply, and in that water can be conveyed by gravity. The project will be an important and basic facility for mitigating the existing crisis of water resources in North China. The first round of channel construction has been launched and will be completed by 2010. The WRP will divert water from the upper reach of Yangtze River into Huang River. Following the economic and technical feasibility studies, this route is expected to be constructed only sometime after (MWR, 2002).

If the South-North Water Transfer Scheme is successfully implemented and the total capacity fully realized as planned, the situation will be fairly positive (Table 2.4). Water shortage will be 10 km^3/yr by 2030 and only 0.5 km^3/yr by 2050.

Table 2.4 Water shortage estimations with full water transfer capacity (km^3/yr) (Liu and Chen, 2001)

River	Year	Water Availability*	Water transfer**	Moderate Growth	
				Demands	Shortage
Huang	2030	44.3	8.5	53.5	0.7
	2050	44.8	9.5	54.5	0.2
Huai	2030	73.5	3.9	81.5	4.1
	2050	76.4	7.4	83.9	0.1
Hai&Luan	2030	40.6	8.1	53.9	5.2
	2050	41.8	13.6	55.6	0.2
Total	2030	158.4	20.5	188.9	10
	2050	163	30.5	194	0.5

* including current water transfer capacity ** future water transfer

However, the real situation is hard to anticipate. The water transfer scheme might not be as effective as foreseen and possible negative effects on the whole ecosystem may counteract the benefits associated with. In general, the reliability of the scheme is associated with high uncertainty. For that reason, within this study, only half of the full water transfer capacity is taken to estimate the future water shortage whilst all other variables are constant as Table 2.3 without future water transfer. With half capacity, water shortage will be 20 km^3/yr by 2030 and 16 km^3/yr by 2050 (Table 2.5), which is high. This implies that the water transfer scheme may not solve the whole water problem and there might still be water shortage in North China.

Table 2.5 Sensitivity of water shortages with half water transfer capacity (km^3/yr)

River	Year	Water availability*	Water transfer**	Moderate Growth	
				Demands	Shortage
Huang	2030	44.3	4.3	53.5	5.0
	2050	44.8	4.8	54.5	5.0
Huai	2030	73.5	2.0	81.5	6.1
	2050	76.4	3.7	83.9	3.8
Hai&Luan	2030	40.6	4.1	53.9	9.3
	2050	41.8	6.8	55.6	7.0
Total	2030	158.4	10.3	188.9	20.3
	2050	163	15.3	194.0	15.8

* including current water transfer capacity ** future water transfer

2.4.3 Potential application of desalination in China

Seawater desalination has been studied in many institutes and universities in China since the 1960's, particularly membrane science and technology (People's daily, 2002). Besides, Chinese scientists have recently developed atomic reactors to provide heating to desalinate seawater, by burning used fuel from nuclear power stations under normal pressure. The breakthrough would be an active factor to facilitate developing of seawater desalination especially for cities with severe water shortage. A pilot project using deep-water reactor under normal pressure of 200 megawatts will be established in the coastal city of Yingkou, in which the daily capacity is expected to amount to 80,000 m^3/d. However, the application of desalination in China to date is still limited. The total capacity of seawater desalination so far is about 18000 m^3/d. The biggest plant is located in Dalian city with a capacity of 10000 m^3/d. More desalination plants have been built to treat brackish-, river- and wastewater, often using RO processes. Given the growing stress of water shortage, desalination becomes important to provide additional clean water from brackish water or seawater. For inland water shortage cities, such as those in Hebei and Shandong provinces, wastewater treatment has double benefits: that of reducing the discharge of waste directly into river, as well as providing more water supplies for the cities. In 1999, there was a total wastewater discharge of about 60 billion m^3/yr including industrial and municipal use. Among them, industry accounts for 67% and municipal use 33%. The ratio of treatment of wastewater was only about 14% in 1997 in China (Liu and Chen, 2001). Water shortage is even more serious in the coastal areas with water resources in some industrial cities averaging only 500 m^3 per person. Therefore, there is a great need to develop wastewater desalination in order to reuse the water. As about two thirds of the water is used for industry in southeast coastal cities, seawater desalination, where applicable, should be considered as an alternative to provide water supply.

To evaluate the feasibility of seawater desalination, it is crucial to look at the costs and consumers' affordability. As we analysed before, increasing desalination plants is a potential solution to solve or at least ease water scarcity in China. However, in a market economy, economical feasibility of building a desalination plant is one of the primary questions that should be answered during the feasibility surveys of investing in such a manufacture. Table 2.6 lists current water prices of consumption for some water shortage cities in China.

Table 2.6 Current water prices in water shortage cities ($/$m^3$)

City	Domestic use	Industrial use	Commercial use
Beijing	0.349	0.386	0.386
Tianjin	0.313	0.458	0.602
Shanghai	0.205	0.157	0.181
Shi Jiazhuang	0.133	0.241	0.265
Taiyuan	0.163	0.205	0.301
Datong	0.145	0.193	0.265
Huhehaote	0.133	0.157	0.301
Shenyang	0.169	0.193	0.289
Dalian	0.277	0.386	0.602
Changchun	0.301	0.554	0.554
Ha'erbin	0.217	0.289	0.482
Nanjing	0.229	0.277	0.337
Zhengzhou	0.193	0.217	0.301
Jinan	0.211	0.253	0.361
Yantai	0.181	0.187	0.301
Qingdao	0.157	0.163	0.163
Xi'an	0.181	0.224	0.301
Lanzhou	0.084	0.120	0.139
Average	0.202	0.259	0.341

From http://www.waterchina.com (03/2003)

At present, the major obstacle in applying seawater desalination in China is its price. The table above shows that the current average water price is about 0.20 \$/m^3 for domestic use, 0.26 \$/m^3 for industrial use and 0.34 \$/m^3 for commercial use. Water charges have been kept low for a long time due to the governmental policy. Water is not fully charged based on the actual cost occurred but subsidized by the government. Water prices do not reflect the true value of water in China. Nevertheless, the price of water has increased during these few years. Rising urban incomes and growing public awareness have paved the way for increases in urban water prices and increasing the reuse rate.

Today households pay very little for water compared to their income. For example, in Beijing households paid 250 yuan (\$30) for water in 2002, which accounts for only 2% of the total annual income of 12000 yuan (about \$1446). The prevailing assumption is that households are willing to pay about 3 to 5 percent of their income for access to clean water (WRI, 1996). Obviously, based on this criterion, the affordability of urban residence in China is still high. People have the ability to pay more for water. The State Council recently reported that the price of urban water supply in Beijing would be increased to 6.0 yuan/m^3 (0.72 \$/m^3) by 2005, which reaches the current cost of desalted brackish water. For other water shortage cities in the north, it will take some time for water prices to increase to the level of desalination costs. The South-North Water Transfer Scheme will somehow alleviate water shortage, but will also increase water prices considerably due to the huge investment capital, by at least 0.1\$/m^3. The transferred water will be as expensive as desalted water in the next 15 years. In reality, the problem of water shortage will not be worked out if water is still considered as only a government good. Instead it should also be treated as an economic good. In a transition economy as in China, the government will realize that subsidy to water sector will not be highly beneficial to the nation in the long run. Water is often wasted or used inefficiently due to the low prices and lack of awareness. Instead it should be put into a market where prices are determined by the principle of market economy. The governmental policy is indeed necessary to lead this pricing reform successfully step by step. As water prices increase and desalination costs continue the trend of decline, it will create higher favourable conditions to apply desalination in China in the future.

2.5 Conclusions

Desalination is becoming a solution for water scarcity in a number of arid countries. For the potential application of desalination in China, the following conclusions can be drawn from this study:

1) Improved desalination technologies and accumulated management experiences have been playing important roles to reduce the unit cost of water noticeably over time. To date, the unit cost of desalted water using the MSF process has been reduced ten times since the 1960s. The average present unit cost is about 0.9 $/m^3. RO technology has developed rapidly in recent decades, which makes the costs lower than the MSF process for a moderate capacity. Based on this study, the average unit cost of the RO process has declined to around 0.7 $/m^3, which is very competitive for traditional water resources. The technological innovation will still bring down the cost in the future.

2) Based on the reduction trend of the desalination costs in the world, the unit cost of 1.0 $/m^3 for seawater desalination using the MSF process is suggested for potential applications in China. In addition, a unit cost of 0.6 $/m^3 for brackish and wastewater using the RO process and 1.0 $/m^3 for seawater would be appropriate. As the technology develops, RO would be a favourable choice for both seawater and brackish water desalination in the country.

3) Water demand and supply projections indicate that water shortage will become ever severe within the next 50 years in China. Especially in North China, although taking into account water to be transferred under the scheme, water deficiency is estimated to be 16 km^3/yr in 2050. This amount of water can be potentially provided by application of brackish-, waste- and seawater desalination. Particularly for coastal cities, desalination can provide water for industries that do not have a high requirement of water quality. Desalination is therefore suggested to be a strong potential for eliminating water shortages in the future.

4) To apply desalination in China, the water price is the major obstacle. Current average water price is still lower compared to the costs of desalination. In the country, water is not charged based on the principle of market economy, rather heavily subsided by the government. To eliminate water shortage in the future, water pricing will be an effective economic instrument to conserve water and raise awareness. Governmental policy should facilitate the pricing reforms and step by step fill the gaps between costs of desalted water and actual water prices. In conclusion, desalination can provide reliable

water supply and will be ultimately economically feasible, therefore it is requested to invest in and undertake consistently research on selecting planting sites and brine disposal in the near future. However, one thing should be noticed is that the costs presented here are resulted from simplified models. Thus planning an actual plant under a specific circumstance needs to conduct the final assessment of costs accurately that are based on more substantive information and specific data.

Appendix Calculation of unit costs for RO desalting plants

Country	Total capacity (m^3/d)	Process	Water quality	Contract year	Investment costs ($/$m^3$)	Amortised capital costs ($/$m^3$)	O&M costs ($/$m^3$)	Total cost ($/$m^3$)	Conversion rate	1995 unit cost ($/$m^3$)	Total installed capacity (m^3/d)
USA US	1249	RO	BRACK	1996	1.25	0.257	0.385	0.642	1.03	0.623	6796731
USA US	927	RO	BRACK	1996	0.94	0.260	0.390	0.651	1.03	0.632	6796731
USA US	4360	RO	BRACK	1996	4.24	0.250	0.374	0.624	1.03	0.606	6796731
USA US	4600	RO	PURE	1996	2.95	0.165	0.247	0.411	1.03	0.399	6796731
USA US	872	RO	RIVER	1996	0.71	0.209	0.313	0.522	1.03	0.507	6796731
USA US	3840	RO	WASTE	1996	5.05	0.338	0.506	0.844	1.03	0.819	6796731
Virgin Isl. VI	1022	RO	SEA	1996	2.66	0.668	1.002	1.670	1.03	1.621	6796731
Antilles AN	17000	RO	SEA	1997	39.7	0.599	0.899	1.498	1.053	1.423	7560204
Australia AU	600	RO	BRACK	1997	0.62	0.265	0.398	0.663	1.053	0.630	7560204
Austria AT	5280	RO	BRACK	1997	4.64	0.226	0.338	0.564	1.053	0.535	7560204
Bahamas BS	600	RO	SEA	1997	1.59	0.680	1.020	1.700	1.053	1.615	7560204
Brazil BR	2400	RO	BRACK	1997	2.41	0.258	0.387	0.644	1.053	0.612	7560204
Canada CA	11000	RO	BRACK	1997	9.6	0.224	0.336	0.560	1.053	0.532	7560204
Cayman Isl. KY	1135	RO	SEA	1997	2.94	0.665	0.997	1.662	1.053	1.578	7560204
Cayman Isl. KY	1600	RO	SEA	1997	1.8	0.289	0.433	0.722	1.053	0.686	7560204
Chile CL	18000	RO	BRACK	1997	16.22	0.231	0.347	0.578	1.053	0.549	7560204
Chile CL	684	RO	SEA	1997	1.8	0.675	1.013	1.689	1.053	1.604	7560204
China CN	5700	RO	PURE	1997	4.11	0.185	0.278	0.463	1.053	0.439	7560204
China CN	5700	RO	PURE	1997	4.11	0.185	0.278	0.463	1.053	0.439	7560204
China CN	10950	RO	PURE	1997	7.9	0.185	0.278	0.463	1.053	0.440	7560204
China CN	15000	RO	RIVER	1997	10.64	0.182	0.273	0.455	1.053	0.432	7560204
China CN	1000	RO	RIVER	1997	0.81	0.208	0.312	0.520	1.053	0.494	7560204
China CN	3600	RO	BRACK	1997	3.62	0.258	0.387	0.645	1.053	0.613	7560204
Colombia CO	1600	RO	BRACK	1997	1.59	0.255	0.383	0.638	1.053	0.606	7560204
Cyprus CY	40000	RO	SEA	1997	91.21	0.585	0.878	1.463	1.053	1.389	7560204
Egypt EG	3600	RO	SEA	1997	9.31	0.664	0.996	1.659	1.053	1.576	7560204
Egypt EG	2500	RO	WASTE	1997	3.44	0.353	0.530	0.883	1.053	0.838	7560204
Egypt EG	4000	RO	WASTE	1997	5.23	0.336	0.503	0.839	1.053	0.797	7560204

Chapter 3

Evaluating the costs of desalination and water transport[2]

3.1 Introduction

Water is a crucial resource for survival and growth of life, as well as sustaining the environment. However, the vast majority of water on the earth is too salty for human use. Ninety-seven percent of the earth's water is found in the oceans, with a salt content of more than 30,000 milligrams per litre (mg/l) (Gleick, 2000). Water, with a dissolved solids (salt) content below about 1000 mg/l, is considered acceptable for a community water supply (IDA, 2000). Because of the potentially unlimited availability of seawater, people have made great efforts to try to develop feasible and cheap desalting technologies for converting salty water to fresh water.

A variety of desalting technologies has been developed over the years, including primarily thermal and membrane processes. The main thermal processes include multi-stage flash evaporation (MSF), multiple effect evaporation (ME), and vapour compression (VC). The membrane processes contain reverse osmosis (RO), electrodialysis (ED) and nanofiltration (NF). The MSF and RO processes dominate the market for both seawater and brackish water desalination, sharing about 88% of the total installed capacity (IDA, 2002)(Figure 3.1). Raw water with different qualities has been treated in desalting plants, dominated by seawater and brackish water (IDA, 2002)(Figure 3.2). Seawater is desalted often by various thermal processes and also by RO, whereas brackish water is treated by means of mainly RO and ED.

Desalination of brackish and seawater has been expanding rapidly in recent decades, primarily to provide water for municipal and industrial uses in arid, semi-arid or water-short areas. It is driven by water stress generated from limited water resources and ever growing demands for water. Continuous progress in desalination technology makes it a prime, if not the only, candidate for alleviating severe water shortages across the globe

[2] Chapter 3 is published as Zhou, Y. and Tol, R.S.J. (2005), Evaluating the Costs of Desalination and Water Transport, *Water Resources Research*, 41(3), Art. No. W03003.

(Ettouney et al., 2002). The market is also driven by the falling costs of desalination, which are due to the technological advances in the desalination process (Tsiourtis, 2001). Till 2002 over 15,000 industrial scale desalination units, with a total capacity of 32.4 million m^3/d, had been installed or contracted worldwide. Among them, non-seawater desalination plants contributed with 13.3 million m^3/d, whilst the capacity of the seawater desalination plants reached 19.1 million m^3/d (IDA, 2002).

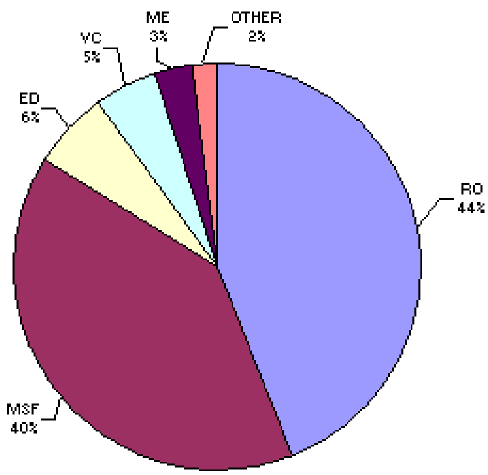


Figure 3.1 Installed desalting capacity by process.

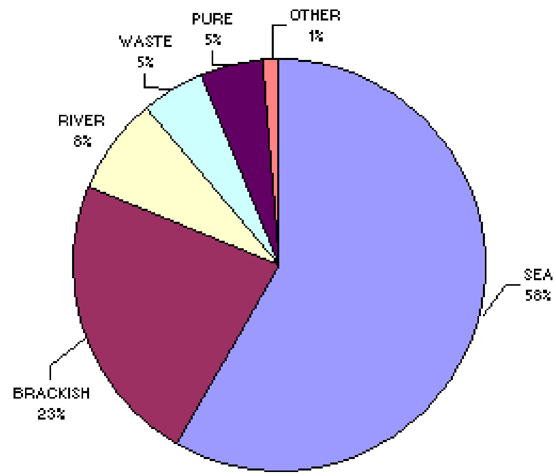


Figure 3.2 Installed capacity by raw water quality.

The costs of water produced by desalination have dropped considerably over the years as a result of reductions in price of equipment, reductions in power consumption and advances in system design and operating experiences. As the conventional water supply tends to be more expensive due to over-exploitation of aquifers and increasing contaminated water resources, desalted water becomes a viable alternative water source. Desalination costs are competitive with the operation and maintenance costs of long-distance water transport system (Ettouney et al., 2002). This study defines the main economic parameters used in estimation of desalination costs and calculates the unit costs of desalted water for five main processes based on simplified assumptions. It then uses multiple regressions to estimate the trends of unit costs over time and analyse the significant factors that affect the cost of desalination. Moreover, in this study a literature survey on the costs of water transport is conducted in order to estimate the total cost of desalination and the transport of desalinated water to where water is short.

3.2 An overview of desalination costs by various processes

The costs of desalination vary significantly depending on the size and type of the desalination plant, the source and quality of incoming feed water, the plant location, site conditions, qualified labour, energy costs and plant lifetime. Lower feed water salinity requires less power consumption and dosing of antiscale chemicals. Larger plant capacity reduces the unit cost of water due to economies of scale. Lower energy costs and longer plant period reduce unit product water cost.

The primary elements of desalination costs are capital cost and annual running cost. The capital cost includes the purchase cost of major equipment, auxiliary equipment, land, construction, management overheads, contingency costs etc. The capital costs for seawater desalination plants have decreased over the years due to the ongoing development of processes, components and materials. Annual running costs consist of costs for energy, labour, chemicals, consumables and spare parts. A typical breakdown of running costs for thermal processes is that the ratio of energy: chemicals: labour equals 0.87:0.05:0.08 (IDA, 2002). The energy costs play a dominant role for thermal processes. Distillation costs will fluctuate more than RO with changing energy costs. In regions where the energy is fairly expensive, RO is a better choice than any other thermal processes due to its lower energy consumption.

To provide the overview of the desalination costs worldwide, we evaluate the unit costs for the main processes based on rough assumptions. All the plants rated at 600 m^3/d per unit or more for the five main processes in IDA Worldwide Desalting Plants Inventory Report No.17 (2002) are included in the calculation. The report provides information on land-based desalting plants rated at more than 100 m^3/d per unit and contracted, delivered or under construction as of the end of 2001. The report is considered to be the most comprehensive and complete of its kind worldwide though not high quality especially in providing more detailed information on single plant. The dataset should be handled with caution since there are no other dataset available to cross check on it. The data regarding desalting plants include country, location, total capacity, units, process, equipment, water quality, user, contract year and investment costs. The detailed annual running costs are not available for the plants so it is hard to differentiate what kind of costs exactly are included and how. The total costs are assumed to be split up into 40% capital costs for interest and depreciation on the investment and 60% of running costs referring to IDA (2002). The load factor is assumed to be 90% for all the plants. These assumptions are the same for all desalination techniques, again for want of better information. We use the IDA (2002) despite the crudeness of the data. The alternative would be to build our own database that may have higher quality and more detailed cost data, but which would also have a much smaller number of observations, have a more limited geographic scope, and cover a much shorter period of time.

The annual amortised capital costs are obtained by multiplying the costs by an amortization factor, given as follows:

$$C = P \times i \times (1+i)^{n-1} \Big/ \left[(1+i)^n - 1 \right] \tag{3.1}$$

where C is amortised annual capital cost, P the investment in the original year, i the annual discount rate, and n the economic plant life. In this study, a discount rate of 8% and a plant life of 30 years are applied for amortization for all the cases. For the purpose of comparison, all unit costs are given in terms of 1995 US dollars calculated based on the United States Consumer Price Index. The cost data and our calculation are available on the web http://www.uni-hamburg.de/Wiss/FB/15/Sustainability/Models.htm.

3.2.1 Costs of the MSF process

This study considers 442 desalting plants using MSF processes worldwide from year 1957 to 2001, with a total capacity of 12.6 million m^3/d. The process accounts for the second largest installed desalting capacity in the world next to RO. The major consumers of MSF are in the Middle Eastern and North African (ME&NA) countries, such as Saudi Arabia, United Arab Emirates, Kuwait, Libya and Iran. The main users of desalinated water are municipality, industry and power plants. The majority of plants are designed to treat seawater.

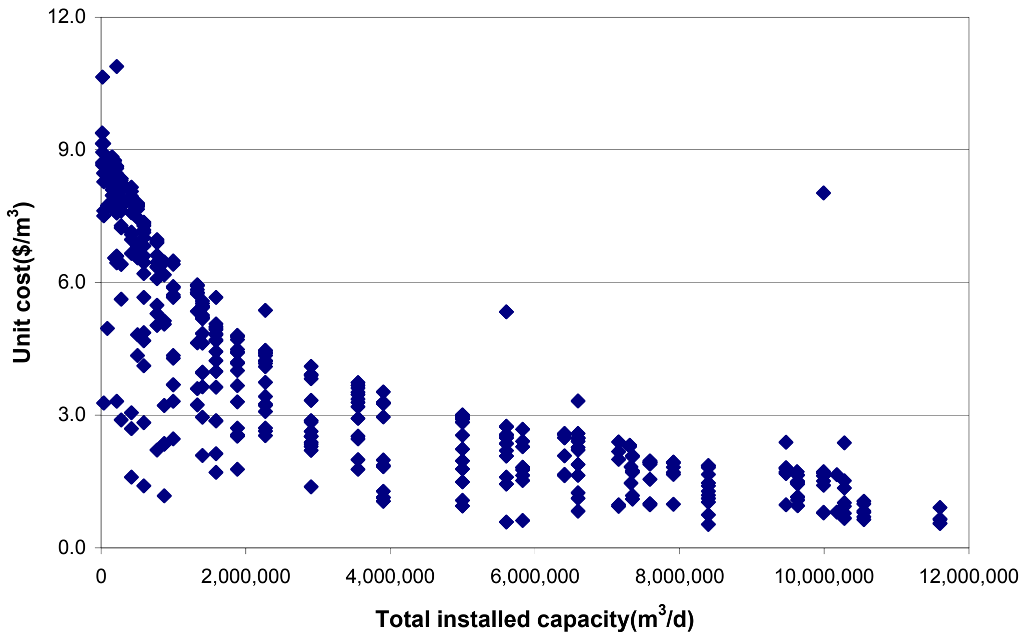


Figure 3.3 Unit costs vs. total installed capacity by the MSF process.

Figure 3.3 illustrates the unit costs of all the desalting plants using the MSF process over the total cumulative installed capacity. The unit cost has been reduced substantially since the initial stage of MSF technology. The average unit cost has fallen from about 9.0 $\$/m^3$ in 1960 to about 1.0 $\$/m^3$ at present, which indicates that there has been a great improvement of MSF technology. The average annual reduction rate of unit costs has been about 5.3% in last 40 years.

We use regression methods to estimate the unit costs of these desalting plants. The original data for the plant include the location, the year, the plant capacity and raw water quality. The calculated data include unit costs and the total cumulative installed capacity. The major consumers for MSF are located in the Middle East and North Africa (ME&NA), therefore regional dummies are included to analyse the significance of location differences. The raw water quality dummies are also included. The model for this process is specified in (3.2).

$$F(UNITC) = G(TIC, CAP, YEAR, ME \& NA, SEA) \tag{3.2}$$

where *UNITC* is the average unit cost of desalting one cubic meter of water, *TIC* refers to the total cumulative installed capacity, which reflects the expansion of desalting plants over time. *CAP* is the capacity of a single plant. *YEAR* is the contract year of the plant. *ME&NA* is the regional dummy, and *SEA* is the raw water quality dummy. The model was estimated with OLS for two different equations, namely semi-log and double log. Since *TIC* and *YEAR* are correlated and non-stationary, we estimate separate equations with either (but not both) explanatory variable. *UNITC* cointegrates with both *TIC* and *YEAR*, and *TIC* and *YEAR* cointegrate with each other. Statistical techniques for multi-cointegration have yet to be developed (cf. Banerjee *et al.*, 1993; Chatfield, 2004), except when there is strong prior information (Tol and de Vos, 1998), which we lack in this case. Note that the two alternative regressions have a different interpretation. With *YEAR* as an explanatory variable, costs reductions are due to technological progress *outside* the water desalination industry. In contrast, with *TIC* as an explanatory variable, cost reductions are due to technological progress *inside* the water desalination industry through learning by doing. The estimation results are presented in Table 3.1.

Table 3.1 Unit cost estimation results for MSF

Variable	Log-log	Log-log	Semi-log	Semi-log	Log-log (adjusted)	Log-log (adjusted)
Constant	6.93*	798.76*	1.21*	109.49*	5.83*	672.22*
	(38.96)	(38.73)	(14.85)	(36.65)	(29.11)	(31.36)
TIC	-0.35*		-1.71E-07*		-0.26*	
	(-30.22)		(-33.95)		(-20.21)	
YEAR		-105.02*		-0.06*		-88.33*
		(-38.59)		(-36.31)		(-31.22)
CAP	-0.16*	-0.14*	-2.21E-06*	-2.14E-06*	-0.17*	-0.14*
	(-12.85)	(-13.30)	(-7.93)	(-8.01)	(-12.19)	(-13.24)
ME&NA	0.10*	0.05	-0.06	-0.05	0.21*	0.17*
	(2.76)	(1.54)	(-1.85)	(-1.41)	(4.94)	(5.05)
SEA	0.63*	0.69*	0.73*	0.68*	0.66*	0.73*
	(7.35)	(9.54)	(8.74)	(8.57)	(29.12)	(9.71)
R^2-adj.	0.77	0.84	0.78	0.80	0.64	0.78
F value	369.37	571.16	393.55	445.83	195.08	397.89
Log likelihood	-161.29	-85.85	-150.44	-130.84	-213.46	-102.99
n	442	442	442	442	442	442

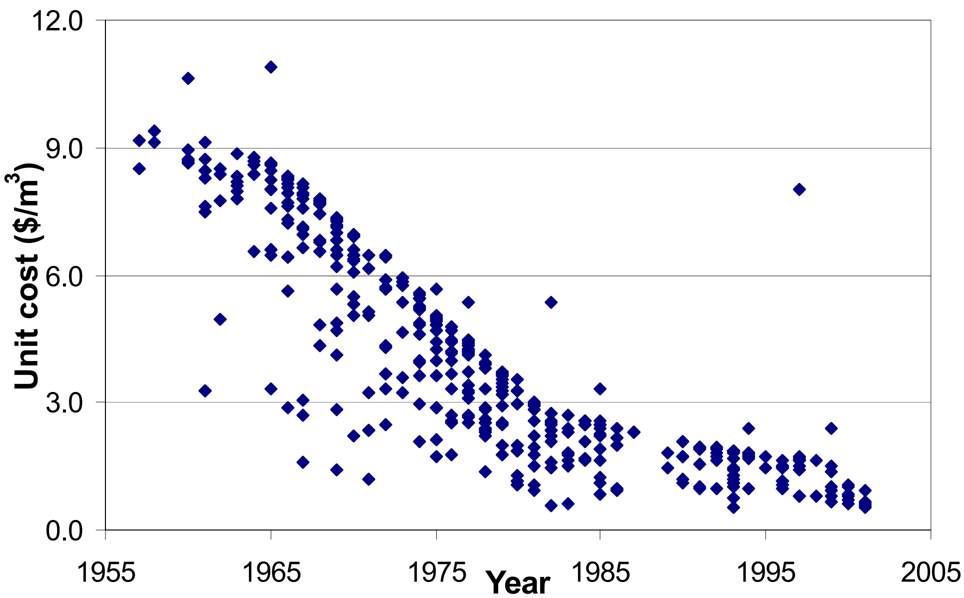
The t statistics are in parentheses.
*Significance at the 0.01 level.

The regressions show that all the variables but *ME&NA* are statistically significant in unit cost estimation. The negative values imply that the unit cost declines with the increase of the variables. As *TIC* represents the total installed capacity of all the desalting plants, the decline of the unit cost can be explained as a result of the technological development and gained experiences. *CAP* also influences the unit cost of a plant, as the cost tends to be lower with the increase of plant capacity due to economies of scale. It is thus suggested from this study that seawater-desalting plants using the MSF process will be economically favourable to have a larger capacity. However, the correlation is not obvious for plants with a capacity less than 50,000 m^3/d (Zhou and Tol, 2004). *YEAR* is significant, reflecting that the technology change outside the sector also plays an important role in the cost reduction over time. The positive value of *SEA* implies the higher unit cost for seawater desalting than for other raw water quality.

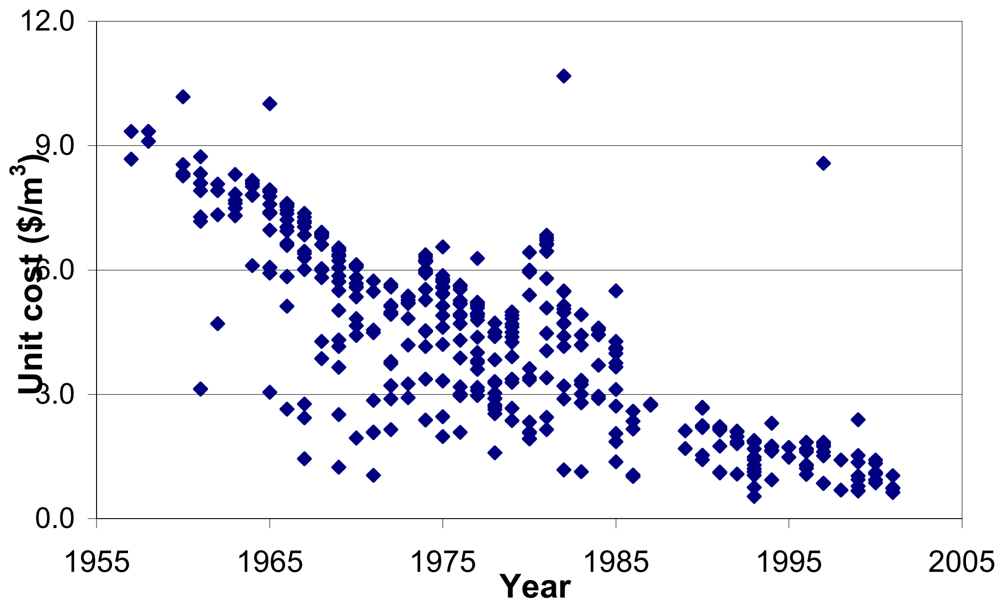
According to the regression results, the unit cost will continue to decrease with the increasing cumulative capacity and over the time. The double log estimation with *TIC* suggests a total installed capacity elasticity of –0.35, that is, for every 1% extension of the total installed capacity, the unit costs decrease by 0.35%. For the year 2001 alone, the total contracted capacity has increased by about 8%. That would mean a decrease of unit cost by 2.8%. The study also indicates an elasticity of –0.16 for the plant capacity, that is, increasing returns to scale.

As energy cost played such a significant role in the total cost of desalination, one may wonder why the curve in Figure 3.3 does not reflect the oil crisis in the 1970's, which had led to the dramatic increase of oil prices. The reason is that the above estimation is conducted irrespective of energy prices due to lack of information on actual energy consumption for all the plants. In order to get an idea of how the energy prices may influence the whole cost of desalination, we report a sensitivity analysis by calculating the unit cost over time based on the correlation between energy costs and oil prices. Although some plants run on natural gas instead of oil, here we take only oil prices since the gas price typically follows the oil price. One may argue that the production costs of plants would not be affected by changes in oil prices because the Middle Eastern countries, where most desalination plants are located, have plenty of cheap oil and gas. However, the market price reflects the "opportunity cost" incurred for not selling oil and gas. The crude oil prices are obtained from the websites of the Office of Transportation Technologies (http://www.ott.doe.gov) and the Energy Information Administration (http//www.eia.doe.gov). We assume that the energy costs account for 50% of the total cost in the year 1995, and then correlate the energy cost in a particular year with oil prices of the time. If the oil price doubles in that year compared to 1995 level, then the energy cost also doubles. Figure 3.4 illustrates the unit costs of MSF plants with and without adjustment for oil prices. Without oil prices, there is a comparatively neater trend than with prices adjustment. Figure 3.4b shows clearly higher costs during the period 1970-1985. Since 1990, the unit costs are more or less similar in 4a and 4b. This analysis indicates that you could expect more or less similar fluctuations of costs for other thermal processes such as ME and VC and perhaps a smaller scale of fluctuations for membrane processes. The regression using log-log model was conducted again with oil prices adjusted data and the result was presented in Table 3.1. Clearly, there is a less correlation for the energy-adjusted data and it also suggests a less total installed capacity elasticity and the significance of plant locations (*ME&NA)*.

Due to the crudeness of data, it is difficult to come up with a realistic analysis of energy costs for all the plants. This analysis is presented here for illustrative purposes only. For the rest of the paper, energy costs are not adjusted particularly with oil prices for desalination cost estimation.



a. Without oil prices



b. With oil prices

Figure 3.4 Sensitivity analyses of unit costs regarding energy costs.

3.2.2 Costs of the RO process

The RO process has become more popular during the past decades due to advancing technology and falling costs. It should be noted, though, that RO plants are more difficult to operate than other types of desalination plants, the main attraction being costs. The operating cost of RO plants has been reduced, thanks to two developments: (1) lower-cost, higher-flux, higher salt-rejecting membranes that can efficiently operate at lower pressures and (2) the use of pressure recovery devices (Gleick, 2000). This study contains 2514 desalting plants using RO processes worldwide, with a total capacity of 12.7 million m^3/d since the 1970's. The process has become to have the largest installed desalting capacity throughout the world. RO is often used to treat less saline water, such as brackish, river and wastewater. Since the last decade, it has been increasingly applied for seawater as well and has become competitive to thermal processes. Till 2001, a breakdown of capacity according to feed water quality is that the ratio of brackish: seawater: river&pure: other is about 40:14:40:6. The users include municipal and industrial use, power plants and also tourism.

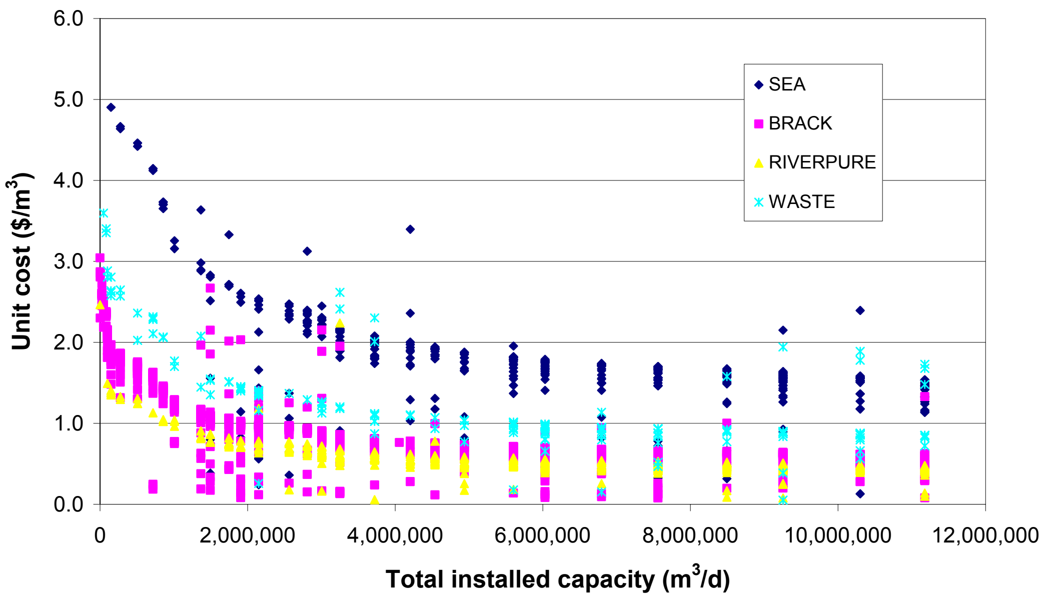


Figure 3.5 Unit costs vs. total installed capacity by the RO process.

Figure 3.5 shows the unit costs of all desalting plants using RO processes over the total cumulative installed capacity. The different feed water qualities are indicated with different symbols. In general, the unit costs for seawater are the highest, followed by waste-, brackish- and river & pure water. Raw water quality plays an important role in the costs of RO desalination. The average unit costs of RO processes have declined from 5.0 $\$/m^3$ in 1970 to less than 1.0 $\$/m^3$ today. Figure 3.5 also shows that the unit costs for seawater desalination are still above 1.0 $\$/m^3$ whilst the cost for desalting brackish-, river- and pure-water has been reduced to less than 0.6 $\$/m^3$ level. Note that recent tenders costs of large seawater RO indicate even lower costs. For instance, some field estimates suggest a cost of $\$0.55/m^3$ for a large RO project in Florida (Ettouney, 2002); more recent cost proposals such as for the Ashkelon desalination in Israel have included costs as low as 0.52 $\$/m^3$ (Busch and Mickols, 2004).

Essentially we did similar regressions to estimate the unit cost as for the MSF process. The major consumers for RO are located quite dispersedly worldwide such as in the USA, Saudi Arabia, Spain, Japan and Korea, which give no information about grouping countries, therefore the regional dummies are excluded. Various raw water qualities such as brackish-, sea-, river-, pure-, wastewater are included. The model specification is in (3.3).

$$F(UNITC) = G(TIC, CAP, YEAR, SEA, BRACK, RIVERPURE) \quad (3.3)$$

where *SEA, BRACK,* and *RIVERPURE* refer to seawater, brackish water and river plus pure water dummies. Wastewater and brine water are in category *OTHER*, which does not show in the equation. The regression results for double-log and semi-log models are presented in Table 3.2.

Table 3.2 Unit cost estimation results for RO

Variable	Log-log	Log-log	Semi-log	Semi-log
Constant	5.19*	652.68*	0.60*	88.66*
	(53.99)	(47.42)	(19.66)	(47.84)
TIC	-0.29*		-9.03E-08*	
	(-50.20)		(-36.85)	
YEAR		-85.81*		-0.04*
		(-47.34)		(-47.77)
CAP	-0.10*	-0.09*	-3.55E-06*	-3.74E-06*
	(-15.85)	(-14.42)	(-6.72)	(-7.89)
SEA	0.50*	0.50*	0.46*	0.49*
	(17.82)	(17.02)	(13.89)	(16.19)
BRACK	-0.41*	-0.42*	-0.38*	-0.41*
	(-16.17)	(-15.89)	(-12.66)	(-15.16)
RIVERPURE	-0.66*	-0.67*	-0.70*	-0.66*
	(-25.17)	(-24.86)	(-22.74)	(-23.76)
R^2-adj.	0.72	0.71	0.62	0.69
F value	1322.63	1216.75	813.19	1122.73
Log likelihood	-639.38	-716.31	-1050.91	-787.04
n	2514	2514	2514	2514

The t statistics are in parentheses.
*Significance at the 0.01 level.

The results show that all the variables are statistically significant at the 0.01 level. The negative coefficient values of *TIC* and *CAP* imply a lower unit cost with the increase of the total installed capacity and the plant capacity, which is similar to the estimation of MSF. Raw water qualities give both positive and negative values. The positive coefficient value of *SEA* implies that there is a higher unit cost for seawater desalination than *OTHER* (wastewater). Negative coefficient values of *BRACK* and *RIVERPURE* indicate a lower unit cost for brackish-, river-, pure- water than wastewater. Moreover, *RIVERPURE* (-0.66) shows a smaller value than *BRACK* (-0.41), which implies that the unit cost of desalting river & pure water is lower relative to that of brackish water. These results make sense in that cleaner and less saline water requires relatively less energy than low quality water in treatment process.

The double-log regression results suggests a total installed capacity elasticity of –0.29, which means that for every 1% extension of the total installed capacity, the unit costs fall by 0.29%. For the year 2001 alone, the total contracted capacity has increased by about 13%, which would mean a fall of unit cost by 3.77%. It also indicates an elasticity of –0.10 for the plant capacity, which is lower than for MSF.

3.2.3 Costs of the ME, VC and ED processes

Three other processes, namely multiple effect evaporation (ME), vapour compression (VC) and electrodialysis (ED), also contribute significantly to desalination. ME and VC are thermal processes applied mainly to seawater desalination whilst ED is a membrane process often used to desalt less saline water. According to IDA Report 17, there are about 143 desalting plants using the ME process worldwide, with a total capacity of 907,000 m^3/d and 289 desalting plants using the VC process with a total capacity of about 1.4 million m^3/d. The VC process was introduced in the 1970s, later than MSF and ME. It was generally used for small and medium scale seawater desalination, but has been developed rapidly in recent decades. In addition, the report comprises 427 desalting plants by the ED process, with a total capacity of 1.3 million m^3/d.

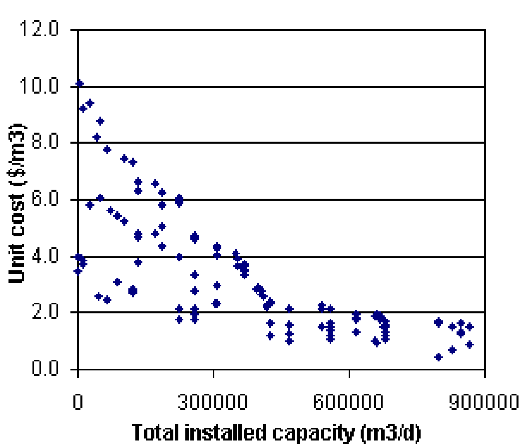


Figure 3.6 Unit costs by the ME process.

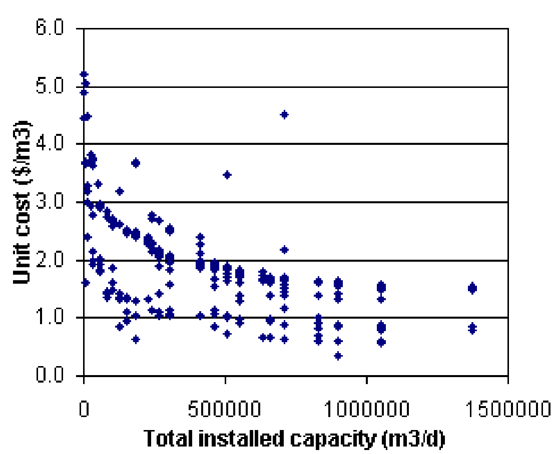


Figure 3.7 Unit costs by the VC process.

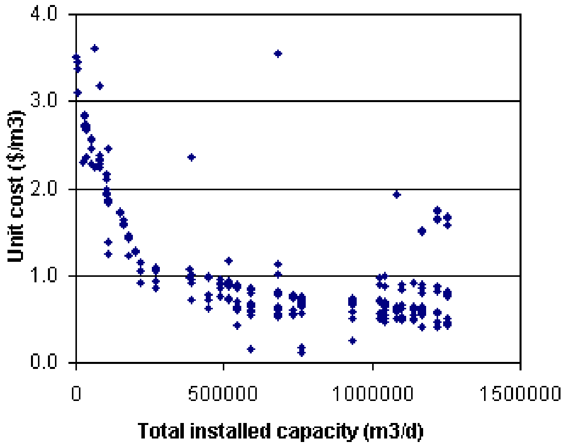


Figure 3.8 Unit costs by the ED process.

Figure 3.6-3.8 show the unit costs of each process over the total installed capacity. The cost of desalination by the ME process has fallen from 10.0 $\$/m^3$ in the 1950's to about 1.0 $\$/m^3$ today. For the VC process, the cost has also decreased considerably over time, from 5.0 $\$/m^3$ in 1970 to about 1.0 $\$/m^3$ at present. As to the ED process, it is remarkable that it has a relatively lower cost than other processes. The average unit cost has gone down from 3.5 $\$/m^3$ in the 1960's to less than 1.0 $\$/m^3$ today. One reason is that brackish water was largely used as feed water. However, the costs seem to go up a bit at the end of the curve, it is because there are a few plants with unknown water quality, which can be wastewater or even seawater. For brackish water desalination, the average unit cost by ED is about 0.6 $\$/m^3$ at present.

Similar regressions were conducted for these three processes as well. Given the dispersed spatial distribution of major consumers, the regional dummies are not included. *SEA* and *OTHER* are included as water quality dummies for the ME and VC processes whilst *BRACK* and *OTHER* are taken for the ED process. The estimation results with double log function for each process are presented in Table 3.3. For ME and VC processes, all the explanatory variables are significant at the 0.01 level. For the ED process, however, it is somewhat surprising that brackish water is not significant, which indicates that the unit costs are independent from raw water quality (excluding seawater). The regression results also suggest an elasticity of the total installed capacity of -0.40 for the ME process, -0.26 for VC, and -0.38 for ED. ME and ED learn faster than MSF and RO and may potentially challenge the two dominant technologies; VC is a slow learner and may never be used for anything but niche applications.

Table 3.3 Unit cost estimation results for ME, VC and ED

Variable	ME (log-log)	Log-log	VC (log-log)	Log-log	ED (log-log)	Log-log
Constant	6.06*	684.72*	4.53*	727.39*	5.42*	677.53*
	(16.26)	(30.80)	(21.35)	(18.20)	(25.08)	(23.88)
TIC	-0.40*		-0.26*		-0.38*	
	(-15.76)		(-17.75)		(-26.87)	
YEAR		-90.06*		-95.59*		-89.15*
		(-30.73)		(-18.16)		(-23.86)
CAP	-0.08*	-0.09*	-0.13*	-0.12*	-0.08*	-0.07*
	(-2.85)	(-4.99)	(-7.04)	(-6.54)	(-5.09)	(-3.91)
SEA	0.74*	0.76*	0.44*	0.39*		
	(9.52)	(16.54)	(11.31)	(9.94)		
BRACK					0.0006	-0.03
					(0.02)	(-0.71)
R^2-adj.	0.67	0.88	0.60	0.61	0.68	0.63
F value	94.86	347.57	146.05	148.97	298.47	240.38
Log likelihood	-52.31	20.34	-44.80	-54.55	-65.80	-99.37
n	142	142	288	288	427	427

The t statistics are in parentheses.
*Significance at the 0.01 level.

To summarise, the unit cost of desalination has fallen considerably since the past 50 years. It was due to the advancing technology in desalination and membrane fields as well as accumulated experiences. The MSF process is still the leading process in seawater desalination, followed by VC and ME processes. The unit cost of desalting seawater has been reduced to about 1.0 $\$/m^3$ or less. RO and ED processes are most often used to treat brackish-, waste- and river water. The unit cost of desalting brackish water has fallen to about 0.6 $\$/m^3$. Due to the lower costs, the expansion of the total capacity of RO plants has been pronounced during the last few years. Particularly for seawater RO, recent tenders have indicated lower costs of large seawater RO plants. RO has shown the great potential to become the most economical process for seawater desalination in the future. As technology and practices grow, the cost of desalination will further decrease.

3.3 Costs of water transport

An extensive search of the scientific literature revealed that little has been published on the costs of transporting water. A few informal interviews with engineers made clear that cost information is held by engineering companies and is considered to be commercially sensitive. The literature search also revealed that most of the few articles that discuss water transport costs refer back to Kally (1993). Kally's 1993 book, however, only sketches the cost estimates, referring for details back to earlier reports in Hebrew. It does contain a few useful estimates, though, particularly with regard to the costs of transferring water from the Nile to Gaza. Our estimates below should be treated with great caution, however.

Transporting 100 million cubic metre (MCM) of water per year over a distance of 200 km would cost 21.4 ¢/m^3. Of this, 4.0 ¢/m^3 are for the purchase of Egyptian water, and 5.2 ¢/m^3 for lifting the water some 100 m. Consequently, it costs 6.1 ¢/m^3 per 100 km to transport water. If the transfer scheme would be extended to 500 MCM, total costs would fall to 19.8 ¢/m^3 and transport costs to 5.3 ¢/m^3 per 100 km. The unit costs of energy and water purchase would not be affected by the extension. This suggests a capacity elasticity of transport cost of 0.92, that is, for every 1% extension of capacity, total costs increase by 0.92% and unit costs fall by 0.08%.

Kally's (1993) cost estimates make clear that horizontal distance is not the main driver of water transport costs, but the vertical distance is. Kally (1993) implicitly makes this point a number of times, but unfortunately does not present cost estimates for alternative lift heights. We therefore assume that the costs of pumping water are linear in the height pumped, in line with Kally's assumption on the energy costs of lifting water.

In his discussion of a possible Red Sea - Dead Sea transfer (for hydropower), Kally (1993) provides the effects of soil type and transfer mode on costs. The Nile-Gaza transfer is by canal in soft but stable soil. If the soil is rocky, transport costs would be 13% higher, and if the soil is sandy, costs would be 175% higher. Transporting water by pipe would lead to a cost increase of 271%, while a tunnel would cost 108% more than a canal.

Gruen (2000) provides estimates of water transport costs from Turkey to Turkish Cyprus. A 78 km pipeline with a capacity of 75 mln m^3 a year would deliver water at 25-34 ¢/m^3. According to Kally's data, the horizontal transport alone would cost 16 ¢/m^3, while effectively lifting the water by 300 m (the sea between Turkey and Cyprus is at least 1000 m deep) would raise the price to 34 ¢/m^3. However, Kally uses an 8% discount rate, while Gruen uses a 4% discount rate; Kally reports that investment and operation and maintenance have an equal share in the costs of transporting water. Correcting for this, Kally's data suggest a cost of some 26 ¢/m^3. The cost estimates of Kally (1993) seem to be consistent with those of Gruen (2000).

Uche *et al.* (2003) report the costs of transporting water in the National Hydrological Plan of Spain. This would involve canals of 900 km long, transporting 1000 mln m^3 of water from the Ebro to Barcelona and Southern Spain. Uche *et al.* (2003) estimate that this can be done at some 36 ¢/m^3 if a 4% discount rate is taken. Based on Kally's data, the horizontal transport alone would cost at least 52 ¢/m^3. Kally's estimates seem to be on the high side.

Hahnemann (2002) discusses the Central Arizona Project, which brings some 1800 mln m^3/yr from the Colorado river to amongst others Phoenix and Tucson, a horizontal distance of some 550 km, and a vertical distance of some 750 m. Kally's data suggest that this would cost some 74 ¢/m^3, but Hahnemann (2002) reports an otherwise unspecified marginal cost of only 5 ¢/m^3.

Liu and Zheng (2002) estimate the costs of transferring water of the Yangtze to China's north. They provide most detail about the eastern route, which is in a more advanced stage of planning than the middle and western routes. The total amount of water transferred is 32 bln m^3/yr, although only less than a fifth of that will reach the final destination. The main canal would be 1150 km long, and the water would need to be pumped 65 m high. Liu and Zheng (2002) estimate the costs at 10-16 ¢/m^3; using Kally's estimates, we find this to be 38 ¢/m^3. However, Liu and Zheng's estimates only include capital; according to Kally, operation and maintenance are of the same order of magnitude as investment costs. Moreover, Liu and Zheng apparently use a zero discount rate, and part of the eastern route uses already existing canals. This suggests that the costs estimated by Liu and Zheng are in fact slightly above Kally's estimates.

In sum, the cost estimates of transporting water by Kally (1993) are the most detailed in the open literature. Comparing these estimates to those of other studies suggests that Kally may have been overly pessimistic. However, most of these studies are *ex ante* engineering studies of government projects, which suggests that the actual costs would have been higher. Therefore, we continue to use Kally's estimates.

3.4 The potential of desalination

Seawater desalination plants are typically located in the coastal area. However, not all the water scarce regions are close to the coast, which generate a need to transport water from desalination plants to where water is needed. In this study, we calculate the total cost comprising the cost of desalination and the cost of transporting desalinated water to the nearest point of distribution. Here we estimate only the cost of source water, not the ultimate costs to the end users. The costs for different end-uses vary according to the system of distribution, blending and purification. For agriculture, the cost is perhaps similar to the cost presented here, but for potable water the cost could be increased as much as 0.1$/m^3 (the cost of additional treatment).

Table 3.4 contains some sample calculations for the costs of desalinated water in selected water-stressed cities. We assume a transport of 100 MCM/yr. Transport costs are assumed to be 6 ¢ per 100 km horizontal transport plus 5 ¢ per 100 m vertical transport. Distances and elevations are taken from the Times Atlas of the World. The calculations are illustrative only.

The costs of desalination, here assumed to equal 100 ¢/m^3, are typically larger than the costs of transport. Indeed, one needs to lift the water by 2000 m, or transport it over more than 1600 km to get transport costs equal to the desalination costs. Thus, desalinated water is only really expensive in place far from the sea, like New Delhi, or in high places, like Mexico City. Desalinated water is also expensive in places that are both somewhat far from the sea and somewhat high, such as Riyadh and Harare. In other places, the dominant cost is desalination, not transport. This leads to relatively low costs in places like Beijing, Bangkok, Zaragoza, Phoenix, and, of course, coastal cities like Tripoli.

Table 3.4 the cost of desalinated water to selected cities

City, country	Distance (km)	Elevation (m)	Transport (c/m^3)	Desalination (c/m^3)	Total (c/m^3)
Beijing, China	135	100	13	100	113
Delhi, India	1050	500	90	100	190
Bangkok, Thailand	30	100	7	100	107
Riyadh, Saudi Arabia	350	750	60	100	160
Harare, Zimbabwe	430	1500	104	100	204
Crateus, Brazil	240	350	33	100	133
Ramallah, Palestina	40	1000	54	100	154
Sana, Yemen	135	2500	138	100	238
Mexico City, Mexico	225	2500	144	100	244
Zaragoza, Spain	163	500	36	100	136
Phoenix, USA	280	320	34	100	134
Tripoli, Libya	0	0	0	100	100

3.5 Conclusions and discussions

In energy-rich, arid and water-scarce regions of the world, desalination is already an important option. As with all new technologies, progress in desalinating water has been rapid. Whereas it costed about 9.0 $\$/m^3$ to desalinate seawater around 1960, the costs are now around 1.0 $\$/m^3$ for the MSF process. For RO, the most popular method, the costs have fallen to 0.6 $\$/m^3$ for brackish water desalination. There is no reason to believe that the trend will not continue in the future. However, it should be noted that the costs of desalination still remain higher than other alternatives for most regions of the world.

Transporting water horizontally is relatively cheap whilst the main cost is lifting it up. We find that desalinated water could be delivered to Bangkok for 1.1 $\$/m^3$, to Phoenix for 1.3 $\$/m^3$ and to Zaragoza for 1.4 $\$/m^3$. These are probably competitive prices at the moment, and they may well fall in the future. However, getting water to New Delhi would cost 1.9 $\$/m^3$, to Harare 2.0 $\$/m^3$, and to Mexico City 2.4 $\$/m^3$. Desalinated water may be a solution for some water-stress regions, but not for places that are poor, deep in the interior of a continent, or at high elevation. Unfortunately, that includes some of the places with biggest water problems.

It should be noted that desalination processes are accompanied by some negative impacts on the environment. The environmental costs associated with desalination – such as production of concentrated brine and carbon dioxide emissions – are not considered in the study due to lack of data. From the literature, the cost of brine disposal is estimated to be 4-5% of the capital cost for a seawater RO plant (Hafez and El-Manharawy, 2002), which is well within the margin of error of our data. In the case of inland brine disposal, brine removal costs can be a more significant portion of desalination costs (10-25%) depending on the circumstances. Therefore, when considering options for massive implementation of desalination, environmental impacts will have to be internalised and to be minimized by proper planning.

In line with desalination, water reuse and recycling are considered and applied increasingly to provide extra usable water. Combining strategies of wastewater reuse and desalination technology makes it possible to convert wastewater into high quality water that suits various users in industry and agriculture. Wherever there is water stress, the improvement of water use efficiencies should be considered in the first place, but its marginal costs should not exceed the marginal costs of enhancing the water supply through desalination.

The analysis presented here provides a general trend of costs under rough assumptions. The selection of most appropriate technology and approach for a particular plant should therefore be based on the careful study of site-specific conditions and economics, as well as local needs. The cost analysis could be improved by having a more detailed and precise running costs for all the desalting plants. For instance, if we know actual energy costs for each plant, the cost estimates would be more realistic. This could be done by collecting a relative smaller amount of plants with high quality data. It would also be interesting to have information on the costs of delivering desalinated water on a geographically explicit basis throughout the world. If we would know the costs of water supply from all other sources for a region, we could then evaluate the potential of desalination. This would require further study in the field.

Chapter 4

Economic analysis of domestic, industrial and agricultural water uses in China[3]

4.1 Introduction

Water is increasingly scarce in many regions and countries. Conflicts over water have involved competition among alternative uses or among geographical regions. In water scarce areas of China, residential and industrial sectors have seized a great amount of water from agriculture and left the land unirrigated. Also, there are often disputes and tensions between upstream and downstream users. As water supply fails to meet the demand in many areas, careful analysis of decisions on the allocation of water is of great significance. The past policy responses to water scarcity are mainly supply-oriented and aim at fostering the development and exploitation of new sources and expansion of the network infrastructure to guarantee the water supply. In recent years water policies have increasingly addressed demand management, which means development of water conservation and management programs to influence water demand. Demand driven measures include adoption of water saving technologies and appliances, awareness raising and economic instruments such as price and tax. The character of water as a scarce good and the need to efficiently price its consumption has gained increasing recognition (Arbues et al., 2003).

The regional variation in availability of water resources in China is considerable, given its diverse climate and geographic conditions. Water is unevenly distributed in both spatial and temporal terms. In South and West China the water endowment is abundant while in the North China Plain water scarcity prevails strikingly. The overall water use in China has grown significantly over the last decades but has levelled off

[3] Chapter 4 is conditionally accepted to publish in *Water Policy*. A modified version is published in Zhou, Y. (2005), *Proceeding of International Conference on Water Economics, Statistics and Finance*, University of Crete, Greece, Book 2, 2-8.

since 1997 (Liu and Chen, 2001). Among the major users, urban and industrial demands for freshwater have grown considerably in recent years. Population growth, rapid urbanisation and overall expansion in economic activities are the major factors underlying the increase in water consumption. Irrigation has been the largest water-using sector although with a decreasing proportion. In addition to water resources, the pattern of water use also differs greatly across regions. For instance, the overall water use per capita ranges from 170 m^3 in water-scarce areas to 2600 m^3 in water-abundant regions (China Water Resources Bulletin 2003).

While the total population in China is projected to stabilize in 2050 and the corresponding water use to level off (Liu and Chen, 2001), the urban population is still expected to increase and the economy likely to continue growing. Irrigation continues to remain dominant in water use but perhaps with high water use efficiencies and a lower quota. Aggregated demand for water is expected to continue its growth. However, developing additional water resources and infrastructure in the future would be limited, as new projects tend to be less profitable and technically harder. Although desalination strikes as an alternative as its cost has decreased significantly in recent years (Zhou and Tol, 2005), it would not be feasible in a few years in China because it is still more expensive than traditional sources and it would take time for desalination to develop and reach the desired capacity which could produce a vast amount of water.

Concerned with the increasing costs of developing new water supply and dealing with the existing inefficiency in the system, an initiative to adopt conservation and water use efficiency measures and a move towards demand management seems urgently needed. In some extremely water-scarce places like Tianjin, water saving technologies and appliances, as well as recycling of water have been widely implemented. At a larger scale, however, demand management measures are still rarely adopted in China. An overview of water uses in various sectors and a better understanding of the determinants help bring forward any effective measure in this direction.

This paper proceeds as follows. Section 4.2 provides a brief introduction to issues related to water demand estimation and a review on previous studies. Section 4.3 describes the data and methodology employed. Section 4.4 elaborates on the regional disparity in the level and pattern of water use, examines domestic, industrial and agricultural uses of water respectively and estimates the corresponding demand by panel data analysis. Section 4.5 provides the policy implications of the analysis, followed by summary and conclusions.

4.2 Water demand estimation issues

Many studies on residential water demand have been done, using various functional forms and data and a range of econometric methods (Espey et al., 1997; and Dalhuisen et al., 2003). The most common explanatory variables include water price, income, climate variables, household characteristics, and the frequency of billing and rate design. Datasets used vary from cross-section, to time-series and pooled panel data. A wide variety of econometric models, ranging from linear, Cobb-Douglas, semi-log to double log, from a single equation to a set of simultaneous equations, have been employed in these studies. The estimation methods used vary as well. The most common one is ordinary least squares (OLS), followed by maximum likelihood (ML). Under simultaneity that occurs often with multiple block rates, instrumental variables techniques, such as two-stage least square (2SLS) or three-stage least square (3SLS) are preferred to OLS estimation (Arbues et al., 2003).

In a meta-analysis of US estimates published between 1967 and 1993, Espey et al. (1997) found an average price elasticity of residential water demand of –0.51. Renwick and Archibald (1998) found that low-income families are more responsive to price (-0.53), whereas the middle and high-income households were less responsive (-0.22). Dalhuisen et al. (2003) also found that the absolute magnitude of price and income elasticities is significantly greater for areas with higher income. Literature survey reveals that little has been published on estimates of price and income elasticities of water for developing countries. Those existing estimates are usually higher than that of developed countries. In the US and Western Europe, the consumption level and service level have reached the threshold of income-inelastic level. Consumers have fulfilled their minimum needs so that an increase in income results in no significant change in water consumption. However, in most developing countries where water infrastructures are obsolete or in dismal situations and the service level is unsatisfactory, income increases are likely to induce the growth in water demand. For instance, people would prefer to improve the sanitary facilities and use water intensive appliances, such as flush toilets, washing machines and dishwashers. The price elasticity of water is also expected to be high because water is under-priced in most regions and a considerable increase in prices is very likely to bring down the current consumption level once consumers start to save water consciously.

Little research has been done on water demand estimation in China. The main reason is the lack of the time-series and cross-sectional data that are typically derived from surveys. There are several urban household water use surveys conducted in municipalities Beijing and Tianjin. The surveys are normally conducted over a short period of one to two years and for a relatively small sample. The survey results are largely presented in qualitative terms regarding the current household characteristics, housing, water using appliances and amenities, water consumption levels, as well as water use behaviour and perception (Zhang and Brown, 2005). The price or income elasticity of water consumption is rarely reported. Cai and Rosegrant (2002) use a price elasticity of domestic water demand of –0.35 to –0.55 that is drawn from previous empirical studies and an income elasticity of 0.75 for China in their global water demand and supply model. Wang and Lall (2002) apply a marginal productivity approach to two thousand Chinese industrial firms to estimate the value of water for industries and suggest an average price elasticity of industrial water demand of –1.0. By analysing the effects of increased water prices on industrial water use in Beijing, Jia and Zhang (2003) suggest a price elasticity of –0.49 for industrial water demand. They conclude, based on an industry survey, that rapid price increases have induced the reduction of industrial water use significantly, especially to those price-elastic industrial firms. In terms of agricultural use, water demand in irrigation has often not been estimated economically due to data constraints and the fact that irrigation charges did not vary significantly until very recently. Nevertheless, under current setting of irrigation institutions, the price elasticity of irrigation water demand is bound to be low and is expected to remain low in the near future (Yang et al., 2003).

4.3 Data and methodology

This study uses aggregated province-level data for 31 provinces or municipalities covering the period 1997-2003. The data on annual precipitation, availability of water resources, overall water use, sectoral breakdown into domestic, industrial and agricultural uses are obtained from China Water Resources Bulletin (CWRB, 1997-2003). The data on population, GDP, urban disposable income, net income of rural households, gross industrial output value, value-added of industry, irrigated land area, value of agricultural production, family size, as well as temperature are derived from the Statistical Yearbook of China (1998-2004). Current water prices for domestic and industrial use are obtained from http://www.waterchina.com. Among these, mean annual temperature and water prices are available only for the capital cities of each province. Therefore the data for each capital city is taken as being representative of the corresponding province. There has been little empirical analysis on water uses of three major sectors in China. This study attempts to bridge the gap by providing a comprehensive analysis on water uses and also for the first time using panel data in examination.

Table 4.1 Summary of the main variables used in the estimations

Variable	Unit	Mean	Standard deviation	Minimum	Maximum
Per capita domestic water use	m^3/person	49.5	20.9	23.9	134
Per capita GDP	yuan*/person	8449	5946	2300	38730
Mean annual temperature	$^{\circ}C$	14.4	5.0	4.6	25.4
Annual precipitation	mm	922.4	559.6	116	2231
Total volume of water resources	billion m^3	91.9	101	0.3	494.6
Water resources per capita	m^3/person	7814	30689	28	196258
Average family size	person	3.6	0.5	2.7	6.8
Domestic water price	yuan/m^3	1.6	0.7	0.7	3.7
Industrial water use	billion m^3	1.8	1.2	0.1	5.8
Value-added of industry	billion yuan	87.3	93.2	0.7	571.8
Industrial water price	yuan/m^3	2.0	1.0	0.9	4.6
Agricultural water use	m^3/hectare	13.1	7.8	3.9	42.4
GDP of primary industry	billion yuan	50.3	36.5	2.9	142.5
Annual net income of rural households	yuan/person	2478	1046	1185	6276

* yuan (Chinese currency, = 0.12 USD)

The data used have a panel structure. The number of period is the same across provinces or municipalities, hence the panel is balanced. The equation for the estimation of water demand is specified as follows:

$$W_{it} = \alpha + \beta X_{it} + \gamma Y_i + v_i + \varepsilon_{it} \tag{4.1}$$

where W_{it} is the dependent variable, X_{it} is the explanatory variables that vary over both time and region, Y_i is the time-invariant variables; v_i is region-specific residual that differs between regions but for any particular region, its value is constant; and ε_{it} is the usual residual that includes non explainable variations that is both spatial and temporal.

β and γ are the slope coefficients associated with the time varying and static variables respectively.

Referring to Wooldridge (2000, 2002) and Green (2003), there are several types of models for panel data and various estimation methods. Equation (1) could be estimated using pooled OLS if the errors are independent, homoskedastic and serially uncorrelated. When there exists unobserved effects and they are not correlated with any element of explanatory variables, we could still apply pooled OLS. However, if they are correlated to any of explanatory variables, then pooled OLS is biased and inconsistent (Wooldridge, 2002). In this case, it is often estimated using the generalised least squares (GLS) in fixed effects and random effects estimations. If the errors are generally heteroskedastic and serially correlated across the time, a feasible generalised least squares (FGLS) analysis can be used. In this study, due to the issues of heteroskedasticity and serial correlation, FGLS was selected to estimate the models.

4.4 Water use and its regional disparity in China

Water use is by definition the amount of water distributed to each sector including the leakage and transfer loss. It is not the final consumption of the users. Water use can be broadly classified into domestic, industrial, agricultural and sometimes ecological uses. The level and purpose of water use differs intrinsically across the sectors. For example, industrial and agricultural sectors uses water mainly as production input as opposed to the residential sector that use water as a direct consumption good. Of the overall water use in 2003, domestic use accounted for 12%, industrial use 22% and agricultural use about 65% (CWRB, 2003). However, the regional disparity in water use is considerable, given the differences in economic activities, social factors, such as culture and customs, as well as the level and pace of economic development. Therefore, it is logical to analyse demand for water in various sectors separately.

Figure 4.1 illustrates the proportional water use of domestic, industrial and agricultural sectors in each province. They do not add to 100% because the rest accounts for ecological use. It shows that agriculture in northwestern provinces, such as Xinjiang, Qinghai, Tibet (Xizang) and Inner Mongolia, has the highest share of water use, which is over 85%. The agricultural use prevails in the South coast regions e.g. Guangxi and Hainan, as well as in the two agriculture-dominated provinces of Hebei and Shandong in the North China Plain. The four municipalities demonstrate the lowest proportion in this case and subsequently higher industrial and domestic uses. In general, moving from east to west across the country, the agricultural use becomes increasingly dominant, while the proportion of domestic and industrial uses fall.

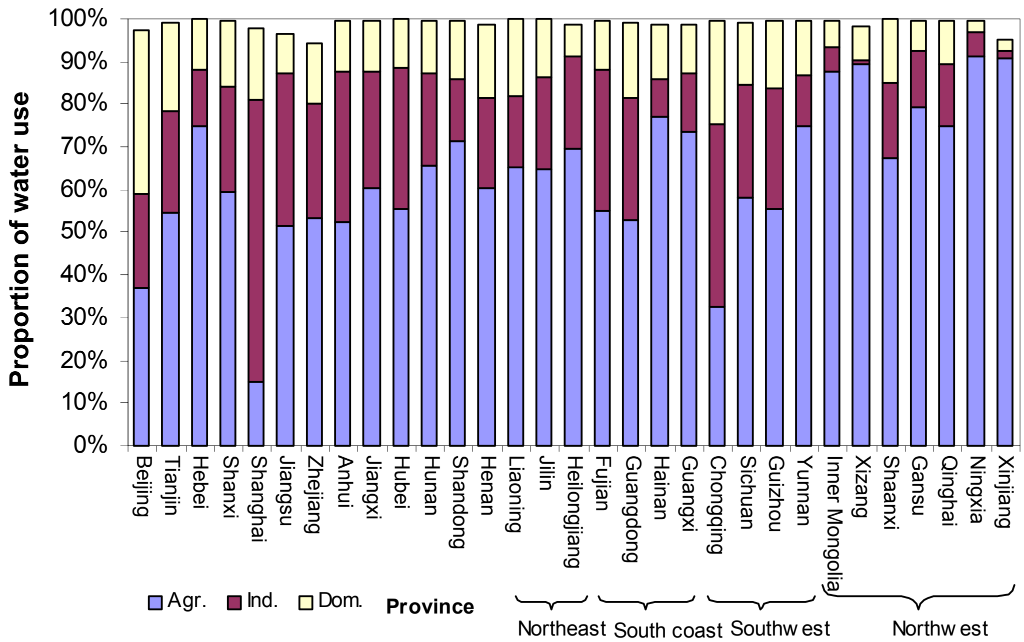


Figure 4.1 Provincial share of water use by domestic, industrial and agricultural sectors

Examining the per capita water use and GDP across the regions, we find that high economic development occurs in the coastal regions of China, while high water consumption appears mostly in the northwest and the South coast regions. It shows little evidence that overall water use depends solely on economic development. Indeed, the economic indicator is only one of the many important determining factors in the level of water use. Figure 4.2 shows the water use per unit value of GDP across the country, which reflects the water use efficiencies in production. It suggests that the under-developed regions use more water for a unit of GDP than developed regions. It is hardly surprising since the poor regions usually have a higher agricultural share in economic activities thus use more water. Agriculture-dominant regions remain economically worse off than industry- or services- dominant economy in China.

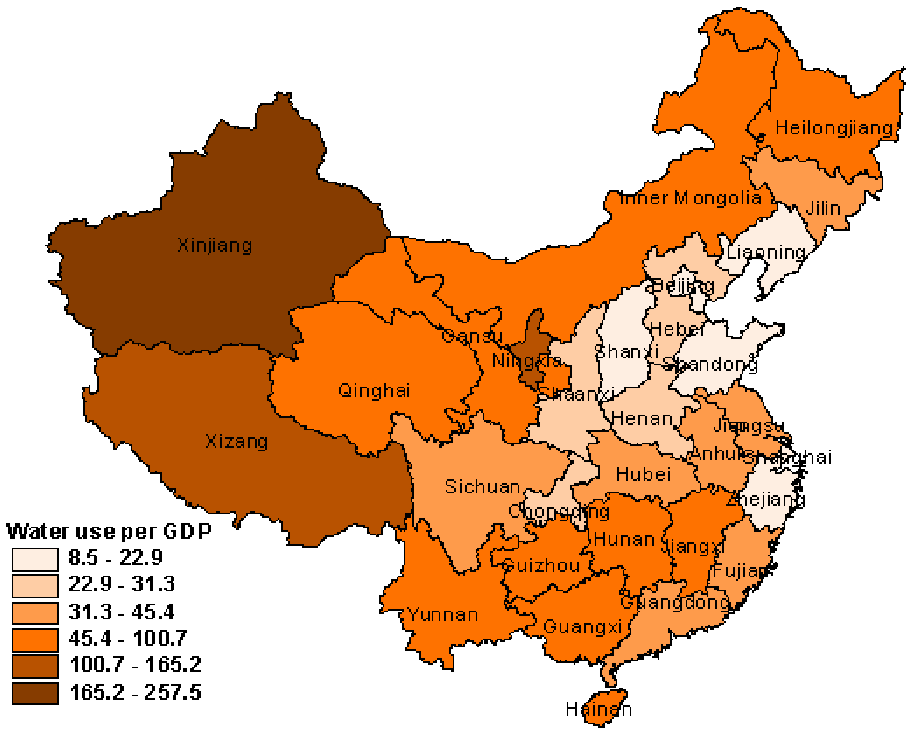


Figure 4.2 Water use per GDP (m^3/1000 yuan) (Data from CWRB, 2003)

4.4.1 Domestic water use

Domestic use here refers to the water used for urban households, urban public sector, rural households and livestock; this follows the definition of the China Water Resources Bulletin. The water consumption for urban households has increased substantially since 1980 with improvements in the standard of living. The migration from rural to urban areas contributes partly to the increase. The rural domestic consumption has also increased but not considerably. The regional disparity in domestic water use is apparent. The average annual domestic water use varies from about 26 m^3/capita in Shanxi province to 107 m^3/capita in Shanghai. Domestic use tends to be higher in wealthy regions such as municipalities Beijing and Shanghai, or water-abundant provinces in West and South China, such as Guangdong, Guangxi and Tibet. Income elasticity of residential water demand measures the rate of response of quantity demand due to a rise (or lowering) in a consumer's income. Price elasticity of water demand is used to measure how consumers change their water consumption in response to changes in prices. We use the panel data to investigate the factors influencing domestic water use. The model is specified as follows:

$$W_d = f(income, prec, temp, wr, famsize, wp, coastal, western) \qquad (4.2)$$

where W_d refers to annual water use of households in a region. GDP per capita is taken as a proxy of domestic income. As domestic use comprises both urban and rural households, a general economic indicator would be more appropriate. GDP value is deflated to its 1997 level using the Consumer Price Index. *Prec* refers to annual precipitation, *temp* is the mean annual temperature, *wr* stands for the average amount of water resources per capita in a region, *famsize* refers to the average family size, and *wp* denotes the average domestic water prices in 2003. Among these, precipitation, temperature, water resources, and household size are panel data, while water prices are cross-sectional data. *Coastal* and *western* are regional dummies, which are included in order to take into account any potential difference between coastal and inland provinces and between western and non western regions. Coastal regions include Beijing, Tianjin, Hebei, Liaoning, Shanghai, Jiangsu, Zhejiang, Fujian, Shandong, Guangdong and Hainan. Western regions consist of Xinjiang, Tibet, Qinghai, Ningxia, Gansu, Shaanxi, Yunnan, Guizhou, Sichuan, Chongqing, and Guangxi.

Due to the panel form of the data set, several tests have to be performed in order to choose the correct specification for the model. To avoid multicollinearity, population density is excluded to be an explanatory variable as it is highly correlated with GDP. The test of the significance of the unobserved effects suggests that the data have a statistically significant group effect but there does not appear to be a significant period effect. Tests for both heteroskedasticity and first-order auto correlation are rejected at 1% level, which implies that the data sets are serially correlated and not homoskedastic. This invalidates the use of OLS for estimation. Following our tests, we use the feasible generalised least square (FGLS) method as an alternative, which allows for the estimation in the presence of AR (1) autocorrelation within panels and heteroskedasticity across panels. The model with data on all the provinces is estimated with assumptions of heteroskedasticity and autoregression in both linear and double log forms. This is presented as Model 1 in Table 4.2. In addition, the model with restricted provinces is performed excluding three exceptionally rich municipalities: Beijing, Tianjin and Shanghai. This is presented as

Model 2 in Table 4.2. We will focus on analysing the results of double log function. The linear model is shown for comparison only.

Table 4.2 Estimation results for domestic water use

	Model 1		Model 2
Variable	Linear	Log-log	Log-log
Constant	18.88** (2.64)	1.86** (6.92)	2.20** (7.61)
Income	1.66** (5.95)	0.42** (8.79)	0.30** (6.21)
Precipitation	0.003** (2.64)	-0.10* (-2.26)	-0.10* (-2.19)
Temperature	0.63** (3.29)	0.23** (3.67)	0.19** (3.12)
Water resources	0.0002** (4.93)	0.13** (7.21)	0.15** (9.25)
Family size	-0.18 (-0.12)	0.21 (1.37)	0.03 (0.20)
Water price	-0.73 (-0.45)	0.04 (0.80)	-0.01 (-0.27)
Coastal	4.91 (1.84)	0.16** (3.54)	0.15** (3.31)
Western	1.81 (0.99)	-0.01 (-0.32)	-0.04 (0.85)
N	217	217	196
Log likelihood	-641.52	160.55	159.64
Wald chi2 (8)	129.15	303.11	276.18
Prob> chi2	0.00	0.00	0.00

The t statistics are in parentheses.
*Significance at the 0.05 level ** Significance at the 0.01 level
Model 1 is estimated for all 31 provinces. Model 2 excludes Beijing, Shanghai and Tianjin.

Most of the coefficients of the explanatory variables are statistically significant and have expected signs. The positive coefficient of income implies that consumers who have a high income tend to consume more water. The positive value of temperature suggests domestic consumers use more water when the weather is relatively warm. Similarly, in water abundant areas more water will be used. Precipitation contributes negatively to water consumption, meaning that households tend to use less water when there is enough rainfall. *Coastal* shows a positive sign, which implies that the coastal regions consume more water than inland regions, which further confirms that richer households tend to consume more water. Family size and water price are not significant at any level, which may be due to the fact that both variables vary little with time.

The double log estimation suggests an income elasticity of 0.42 in model 1, that is, for every 1% increase of domestic income, the domestic water use increases by 0.42%. Model 2 suggests a somewhat lower income elasticity of 0.30. The difference in the two values suggests a high sensitivity of the income elasticity to the data used. Both values are comparatively lower than the presumed value of 0.75 in Cai and Rosegrant (2002). As China continues to develop at a rapid pace and domestic income consequently increases, the water demand is thus expected to grow. The estimation also suggests a precipitation elasticity of –0.10 and a temperature elasticity of about 0.23. Due to the lack of time-series water prices, this study fails to provide a valid estimation of price elasticity of water. The estimate in Table 4.2 is not significantly different from zero. Also, lacking monthly data, the seasonal differences cannot be captured by this analysis.

Table 4.3 Double log estimation results for split samples

	Sample 1		Sample 2	
Variable	Rich	Poor	Water rich	Water scarce
Constant	0.51(1.32)	2.33** (7.68)	1.59** (4.94)	3.49** (5.41)
Income	0.60**(9.43)	0.21** (3.63)	0.37** (8.69)	0.40** (5.13)
Precipitation	0.05 (0.78)	-0.19** (-4.21)	0.04 (0.77)	-0.12 (-1.70)
Temperature	0.08 (1.03)	0.42** (4.55)	0.10 (1.20)	0.10 (0.79)
Water resources	0.02 (0.72)	0.17** (11.62)	0.07** (2.75)	0.03 (0.96)
Family size	1.17**(4.29)	-0.10 (-0.72)	0.44* (2.49)	-0.36 (-0.97)
Water prices	-0.05 (-0.68)	-0.03 (-0.52)	-0.15** (-2.56)	0.15 (1.63)
N	112	105	119	98
Wald chi2 (6)	176.71	205.49	111.57	104.29
Prob> chi2	0.00	0.00	0.00	0.00

The t statistics are in parentheses.
*Significance at the 0.05 level ** Significance at the 0.01 level

In addition, the data were further split in two ways: rich provinces versus poor provinces, and water abundant provinces versus water scarce provinces. The rich versus poor provinces is split based on the median of the per capita GDP. The water abundant versus scarce provinces is split according to the median of the available water resources per capita. They are shown respectively as Sample 1 and 2 in Table 4.3. The results in Sample 1 suggest a large difference between rich provinces and poor ones. The estimation with rich provinces indicates an income elasticity of 0.60, which is higher than the average in Table 4.1. Family size is significant in rich provinces, but not in poor ones nor in the whole sample. However, the estimation of poor provinces suggests a much lower income elasticity of 0.21 but a negative significance of precipitation and positive significance of temperature and water resources. The difference between income elasticity implies that rich households are more responsive to income changes than poor ones in terms of water demand. It may be explained by that households in poor provinces prefer to use the increase of income to meet other needs such as food or shelter than water. Alternatively, the reason may be that, in richer provinces, there are water-intensive household appliances (flush toilets, washing machines) while in poorer provinces these are largely absent. Table 4.3 also shows that poor provinces are more responsive to natural and climatic conditions than richer ones. Surprisingly, the results in Sample 2 indicate that water abundant regions are more responsive to availability of water resources and water prices while water scarce regions only respond to income.

4.4.2 Industrial water use

Industrial use refers to the amount of water withdrawn for industrial purposes, excluding recycling water in firms. The major water consumers are metallurgy, timber processing, paper and pulp, petroleum and chemical industries. Industrial water use presently accounts for 22% of the overall water use in contrast to 10% in 1980. Yet the total volume of industrial water use has stopped growing in recent years. Jia et al. (2003) use the Environmental Kuznets Curve to analyse the relationship between industrial water use and economic development, drawing on the experiences of developed countries. They conclude that industrial water use increases up to a capita GDP threshold in the range of US$3700-$17000 (Purchasing power parity, base year of 1985) and decreases thereafter. The corresponding secondary industry share in the total GDP is 30%-50%. According to this, about half of the regions in China have reached this criteria therefore a drop in industrial water use is expected. Improvement in water use efficiencies is the primary factor for reducing industrial water use, coupled by economic structure adjustment that includes moving from conventional heavy industries towards high-tech and knowledge-based industries. The main driving incentives are the pressing need for upgrading of the industrial structure, more stringent environmental laws and regulations, as well as cutting down the costs for potential resources or environmental crisis. The actual water use per production value has declined rapidly, thanks to the economic structure shift and an improvement in water use efficiencies.

It is straightforward that industrial water use depends highly on the magnitude of industrial firms, particularly of those water intensive industries. If the industrial structure of a region consists of a high proportion of water intensive industries, it is most likely that it has a high water use. However, our concern is not the total amount of water used by industry, but the water use per industrial production value, which reflects the water use efficiency of the industrial sector. Therefore, the model is specified as follows:

$$W_i = f(outp, prec, temp, wr, wp, coastal, western) \quad (4.3)$$

where W_i stands for water use per gross industrial output value, which reflects the sectoral water use efficiency in a region. *outp* refers to the value-added of the industry, and its value is deflated to the 1997 level. *wr* is the total amount of water availability in a region, and *wp* denotes the average industrial water prices in 2003. All the other variables are defined as before.

Table 4.4 Estimation results for industrial water use

	Model 1		Model 2
Variable	Linear	Log-log	Log-log
Constant	1.82** (9.86)	1.02** (3.15)	1.31** (3.97)
Output	-0.25E-03** (-8.52)	-0.32** (-10.35)	-0.38** (-11.0)
Precipitation	0.68E-04 (1.07)	-0.02 (-0.27)	-0.06 (-0.88)
Temperature	0.05** (4.88)	0.63** (6.2)	0.74** (7.13)
Water resources	0.001* (1.96)	0.10** (3.26)	0.11** (3.01)
Water prices	-0.26** (-9.22)	-0.35** (-4.51)	-0.22** (-2.74)
Coastal	-0.78** (-7.18)	-0.66** (-9.00)	-0.63** (-7.87)
Western	0.001 (0.01)	-0.22** (-2.81)	-0.28** (-3.36)
N	217	217	196
Log likelihood	-7.35	77.78	88.51
Wald chi2 (8)	515.79	462.16	364.44
Prob> chi2	0.00	0.00	0.00

The t statistics are in parentheses.
*Significance at the 0.05 level ** Significance at the 0.01 level
Model 1 is estimated for all 31 provinces. Model 2 excludes Beijing, Shanghai and Tianjin.

The model is estimated using FGLS, with assumptions of heteroskedasticity and autoregression. The model estimates with all data and a subsample are shown in Table 4.4. The regression suggests that there is no correlation between precipitation and industrial water use. Most other variables are statistically significant and have the expected signs. The output of industry has negative signs, implying that when the value of industrial production grows in a region the water use per production value declines. This makes sense because through learning by doing and technological change industries tend to improve water use efficiencies. Additionally, industrial structural change helps to phase out water intensive industries. Water price also has negative sign, which means that an increase in industrial water price will result in a decrease in water use. Temperature and the amount of water availability contribute positively to industrial water use. In addition, coastal regions have a lower water use per production value than inland regions.

The regression results suggest an output elasticity of water use of –0.32, i.e. for every 1% increase of the output of industry the industrial water use (per unit of output) decreases by 0.32%. The estimation also suggests a water price elasticity of – 0.35 with all data and –0.22 with a subsample, which are a bit lower than previous estimates of -1.0 (Wang and Lall, 2002) and –0.49 (Jia and Zhang, 2003). Again, the value of price elasticity should be taken with caution due to the problematic data.

We did not do a split sample analysis, because one would want to split the sample according to industrial classification. Unfortunately, we lack the data for this.

4.4.3 Agricultural water use

Agricultural use comprises of water used for farmland irrigation, forestry, animal husbandry and fishery. Although the total amount of water used in agriculture has not increased in recent years, it remains the largest water user. The proportion of agricultural water use has decreased from 83% in 1980 to 65% in 2003 as a result of giving priority to growing population and expanded industry, as well as increasing scarcity of water. The regional variation of agricultural products, practices and water use in China is rather wide. For example, South coast (e.g. Hainan, Guangdong, Fujian) and Northwest (e.g. Xinjiang, Ningxia and Qinghai) have the highest water use per irrigated land area. The South coast also demonstrates higher grain productivity to water input. In contrast, the Northwest consumes vast amount of water but shows the lowest productivity to water input, which is less than one sixth of the national average. The Southwest (e.g. Sichuan, Guizhou, Yunnan) demonstrates the highest productivity, for example, the grain productivity of Sichuan province is about 2.4 times higher than the national average (Kaneto et al., 2004).

Water leakage in irrigation systems and networks is one of the major sources of inefficiency. Bringing water to the field also involves substantial waste of water. In general, the average water use efficiency is between 0.4-0.6 in irrigation ditch, 0.6-0.7 in the field and about 0.5 for irrigated water (Liu and He, 2001). Flood irrigation is predominant and more advanced technologies, such as sprinkle and drip irrigation are not widely practiced. A survey in northern China reveals that other more efficient and yet less capital and energy intensive water saving methods, such as canal lining, border irrigation, hose water conveyance and water quantity and timing control are also not widely used in the region (Yang et al., 2003). The production benefit of farm water (crop yield per unit water quantity) is less than 1 kg/m^3.

The common inputs for agricultural production in empirical analyses are land, labour, chemical fertiliser and agricultural machine. Kaneto et al. (2004) also take water into account as an input to analyse the water efficiency in agricultural production in China. In this study rather than taking a production function approach we focus on examining the direct relationship between agricultural water use and influential factors. If we consider the total agricultural use, it is hard to compare between regions because of the variation in the cultivated area. Therefore, water use per irrigated land area is more appropriate to be the dependable variable. The model for agricultural water use is specified as follows:

$$W_a = f(agrp, prec, prec_{t-1}, temp, income, famsize, coastal, western) \qquad (4.4)$$

where W_a refers to the agricultural water use per hectare, the GDP value of primary industry (mainly agriculture) is taken to be a proxy for agricultural production, and *income* stands for per capita net income of rural households in a region. $prec_{t-1}$ refers to precipitation in the previous year. All other variables are as defined before. All the monetary variables are deflated to the 1997 level. Irrigation water price is believed to be considerably low in China. Water fees are not usually charged on a volumetric basis in rural areas, but levied according to the land area. However, we do not have access to the price (or fees) for each province thus it is not included. The model is estimated using FGLS and the results are shown in Table 4.5.

All the explanatory variables except precipitation are statistically significant. Agricultural production has a negative sign, which implies that water use per land area decreases where agriculture is widely practiced or is highly productive. This may reflect that through learning by doing water use efficiency increases; it may also be because of increasing returns to scale. However, as the agricultural products pertain to both rain-fed and irrigated land, the correlation between water use and agricultural products from irrigated land is hard to observe. Mean annual temperature has a positive sign, which means higher amount of water is irrigated per land area in warmer areas. This is consistent with the agronomic information that the elevated temperatures generally facilitate crop growth, which in turn increase the amount of water uptake by crops. Precipitation (t-1) is positively correlated with water use, which implies that sufficient rainfall in the previous year provide more water for the coming year. The positive values of rural net income and family size imply that more water is used for irrigation when farmers are financially better off or have a bigger family. In general, farmers with higher income can afford their own irrigation facilities, such as drilling wells and pumping water in farmland or conveying water from rivers into their farmland if otherwise unavailable. *Western* shows positive sign, implying that western regions irrigate more water per land area compared to other regions. It is somewhat surprising that *coastal* also has positive sign and a greater coefficient in double log estimation.

Table 4.5 Estimation results for agricultural water use

	Model 1		Model 2
Variable	Linear	Log-log	Log-log
Constant	-1.84 (-0.84)	-0.41 (-0.49)	0.51 (0.71)
Agricultural production	-0.007** (-9.71)	-0.24** (-5.89)	-0.22** (-4.77)
Precipitation	-0.98E-03 (-0.34)	0.007 (0.27)	-0.02 (-0.84)
Precipitation (t-1)	0.9E-02** (4.17)	0.06** (2.84)	0.03* (1.95)
Temperature	0.29** (4.27)	0.20* (2.41)	0.17* (2.19)
Rural income	0.003** (6.23)	0.35** (3.69)	0.26** (3.13)
Family size	1.32** (2.70)	0.40* (2.26)	0.41*8 (2.62)
Coastal	1.86* (2.35)	0.27** (3.18)	0.39** (4.41)
Western	2.98** (5.16)	0.23** (3.38)	0.23** (3.26)
N	216	216	195
Log likelihood	-337.34	194.68	218.60
Wald chi2 (7)	255.49	120.64	94.56
Prob> chi2	0.00	0.00	0.00

The t statistics are in parentheses.
*Significance at the 0.05 level ** Significance at the 0.01 level
Model 1 is estimated for all 31 provinces. Model 2 excludes Beijing, Shanghai and Tianjin.

The regression suggests a production elasticity of water use of –0.24, i.e. for every 1% increase of the value of agricultural products, water use per hectare decreases by 0.24%. In contrast, a 1% increase of rural net income will result in a 0.35% increase in water use per hectare. These two factors pull agricultural water use in opposite directions. It also suggests that water use increases by 0.2% for every 1% increase in temperature. Overall, our results show that water use per irrigated land area is significantly influenced by the economic as well as climatic variables.

Table 4.6 Double log estimation results for split samples

	Sample 1		Sample 2	
Variable	Urban	Rural	Water rich	Water scarce
Constant	-4.87** (-3.0)	1.73 (1.80)	-2.91** (-3.84)	-3.50* (-2.17)
Agricultural production	-0.19** (-3.46)	-0.25** (-4.78)	-0.18** (-3.58)	-0.42** (-9.66)
Precipitation	0.07 (1.13)	-0.02 (-0.51)	-0.02 (-0.52)	0.008 (0.18)
Precipitation (t-1)	0.12** (2.58)	0.03 (1.33)	0.06** (3.26)	0.12** (3.06)
Temperature	0.20 (1.54)	0.03 (0.23)	0.09 (0.94)	-0.09 (-0.73)
Rural income	0.64** (3.55)	0.20 (1.67)	0.71** (8.13)	0.84** (6.20)
Family size	1.40** (3.18)	0.37 (1.76)	0.50** (2.73)	0.82 (1.63)
N	104	111	111	104
Log likelihood	49.97	117.59	123.55	59.57
Wald chi2 (6)	104.2	58.2	94.11	147.5
Prob> chi2	0.00	0.00	0.00	0.00

The t statistics are in parentheses.
*Significance at the 0.05 level ** Significance at the 0.01 level

The data were again split into urban versus rural provinces in Sample 1 and water rich versus scarce provinces in Sample 2. The urban versus rural provinces are split based on the proportion of urban and rural population as well as the ratio of primary industry to GDP. The water rich versus scarce provinces are split according to the absolute value of water resources. Sample 1 shows different results between urban and rural provinces. Water use in rural provinces is more responsive to increase in agricultural production than that in urban provinces although the difference is not significant. In urban provinces, water use shows a positive correlation to precipitation in the previous year, rural income and family size. Comparing the results in Sample 2, it is noticeable that agricultural water use in water scarce provinces shows higher output elasticity and it is also more responsive to changes in precipitation in the previous year and rural net income, although these differences are not significant. Nonetheless, the found differences are all intuitive. The lack of significance is probably due to the limited amount of data.

4.5 Policy implications

Water scarcity and its impact on both human welfare and economic activities have significant implications in China. Yet how to deal with the problem remains controversial. Supply-oriented solutions are still the main interest of the government although demand management including water conservation and a rational pricing system becomes increasingly recognized. Generally, demand-oriented measures are not widely adopted. Great efforts need to be made to improve the level of water management in this regard. To optimise the economic efficiency, water will need to be allocated on market principles rather than on administrative planning principles. Resources and service charges for all sectors are not currently set at appropriate levels to reflect the cost of service delivery or the degree of scarcity. Industry and domestic consumers who already have priority to the use of water need to be charged more since the marginal value of water use is high in these sectors. Agricultural use of water needs to be charged on a volume basis where applicable rather than on a per hectare basis. However, water-pricing reform involves relevant institutional and legal supports that are likely to be difficult. Institutional changes include further development in water resources allocation between and within river basins and the various sectors, regulations and enforcement, demand management, financing and incentives, and service delivery organizations.

Water conservation can be achieved in the domestic sector by introducing appliances such as taps, toilets, and showering devices with water saving features, using water pricing instruments e.g. increasing block tariff as well as by awareness raising. In the industrial sector, this can be achieved by increasing recycling of water, using desalinated water, and most importantly, by putting the right price on water and enforcing a charge on wastewater discharge.

There is plenty of room for improvement of water use efficiency in the agricultural sector. Measures to improve irrigation system efficiency for on-farm and main canal systems include water-saving measures and low-yield land improvement. This involves investing in water infrastructure, irrigation system and irrigation equipment as well as providing relevant knowledge and technical support. Under the current set-up of water pricing, farmers have little incentive to invest in irrigation. Also, farmers who live on the marginal financial gain are reluctant to increase any of their expenses in terms of irrigation. Under current irrigation management system, further increasing the irrigation price may not serve the purpose of water conservation (Yang et al., 2003). In this context, significant reform in the agricultural sector will be required. A top down approach will be appropriate to begin with. The government needs to take initiatives in investing in water and irrigation system to improve the water transfer and conveyance efficiencies, and by promoting more advanced irrigation technologies. Following this, a water pricing reform should be carried out aiming at raising the water price to its supply costs. Ideally the collected revenues could again be used to maintain and improve the water system. Water Use Associates are often proposed to replace the irrigation district units in taking charge of maintaining irrigation system and collecting fees for improving efficiencies.

4.6 Summary and discussions

This study presents econometric analyses of domestic, industrial and agricultural use of water in China using province level panel data. It provides insights into the responses of consumers to exogenous changes. By examining the water use patterns, the study concludes that water use varies substantially across various sectors and across regions. Economically developed or more industrialised areas at the coast consume less water than the agriculture dominated economy in provinces of the west and far south of China.

In general, the results of the study indicate that water uses are significantly related to both economic and climatic variables. For the domestic sector, income is the dominant factor influencing the magnitude of water use and shows an income elasticity of 0.42. We find that richer provinces have a *higher* income elasticity than do poorer provinces. Availability of water resources and climatic variables also contributes to domestic water use. The coastal area shows a relatively higher consumption than the inland area. For the industrial sector, water use per industrial output is negatively related to industrial development, which implies that less water is needed when the industry grows and expands. Industrial demand is also responsive to price changes and shows a price elasticity of –0.35. As for the agricultural sector, the water use per irrigated land area decreases when agricultural production increases. Rural net income, family size and temperature contribute to agricultural use positively.

One issue that needs to be discussed is the water prices used in this study. The prices are average water prices that are set by local water authorities. Whether they result from municipal negotiation or based on some kind of expected consumption regime of the region is not clear. They may not reflect the true marginal cost of supply. In addition, the crudeness of data from CWRB should be noted, as they are aggregated measures of water flowing to different sectors. They may not reflect well the real water consumption in each sector. The reason that we still use this source is because that it is the best available dataset for a comprehensive analysis of water demands in China.

The results of this study are of direct relevance to water resources planning and policy making. The estimates of elasticities can be used in water demand forecast or in cost benefit analysis of future water supply projects. The estimation can be improved by using panel data covering a longer time period or more disaggregated sub-regional level analyses. It would also be useful to extend the study with more adequate data especially regarding time series water prices for the various sectors. Well-designed household surveys would provide richer information and greater insights into the factors influencing domestic water demand.

Chapter 5

Valuing the health impacts from particulate air pollution in Tianjin[4]

5.1 Introduction

China faces the challenges of both strengthening its economy and protecting its environment. Over the last decades, the intensified process of industrialisation and urbanisation, 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 (Economist, 2004). More than 80% of the Chinese cities had SO_2 or NO_2 levels above maximum guideline levels set by WHO (Worldwatch, 2005). The current breakneck speed of industrialisation 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 (Balfour, 2005). 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 (Worldwatch, 2005). 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.

[4] Chapter 5 is submitted to *Environmental Modelling & Assessment.*

Research in the past decades confirms that outdoor air pollution contributes to morbidity and mortality (Holgate et al., 1999). 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 (see reviews by Pope et al., 1995a; Pope, 2000; Brunekreef and Holgate, 2002; and Dockery, 2001). 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 (Pope, 2000). 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 characterised. 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 (see Dockery et al, 1993; Dockery, 2001; Samet et al., 2000; and Schwarz et al., 1996). 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 (WHO, 2000).

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 (e.g. US EPA, 1999; Monzon and Guerrero, 2004; and Danielis and Chiabai, 1998). There are also a couple of studies conducted for Shanghai, Liaoning and Shijiazhuang in China (e.g. Kan and Chen, 2004; Xu and Jin, 2003; and Peng et al., 2002). 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 5.2 describes Tianjin municipality and its air quality, and estimates the air pollution level and population exposure. Section 5.3 introduces the methodology and data used in the study. Section 5.4 develops the monetary evaluation of the health impact of air pollution while Section 5.5 concludes the paper.

5.2 Tianjin and its air quality

Tianjin, China's third largest urban area and a major industrial centre, 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 centre of China's Bohai rim and is being built into a modern port city and a major economic centre in North China. Tianjin has a population in excess of 10 million, of which 59% are counted as urban, with most living in the city itself (Tianjin Statistical Yearbook, 2004). 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 utilisation 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 (Chen et al., 2004). Walsh (1998) 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 centre. 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.

Figure 5.1 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 (Tianjin Environmental Quality Bulletin, 2004).

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 μg/m^3 for SO_2 and 40 μg/m^3 for NO_2).

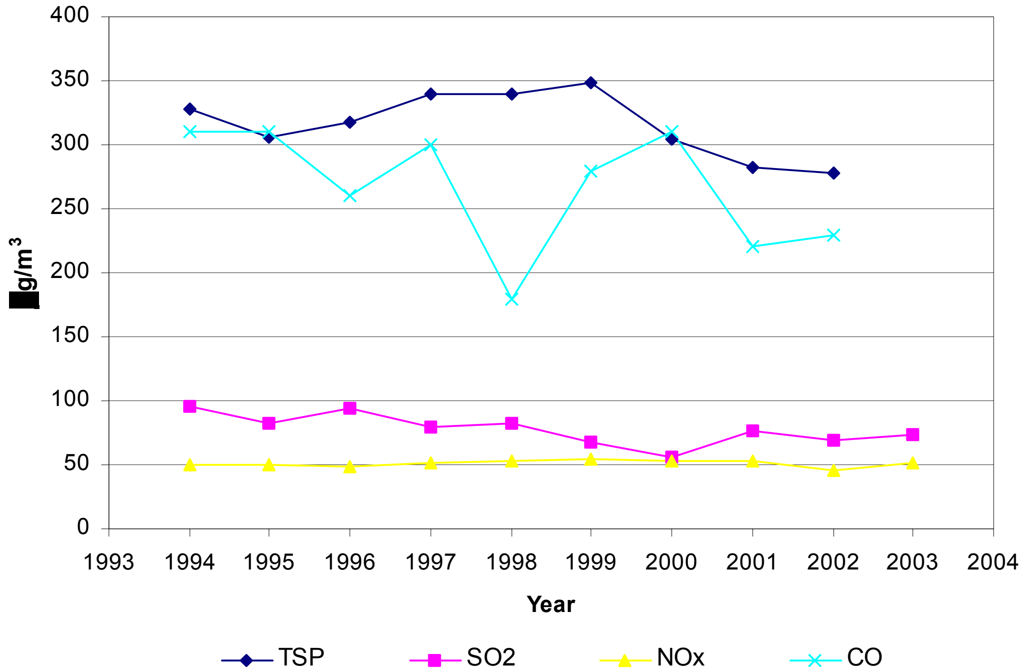


Figure 5.1 Annual average pollutants concentrations in urban area of Tianjin

Air quality is often poor, especially during the winter heating period. The monthly variations in Figure 5.2 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. (2003) 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 Figure 5.2, PM_{10} violated the Class II of the standard (150 μg/m^3) for five months in 2002.

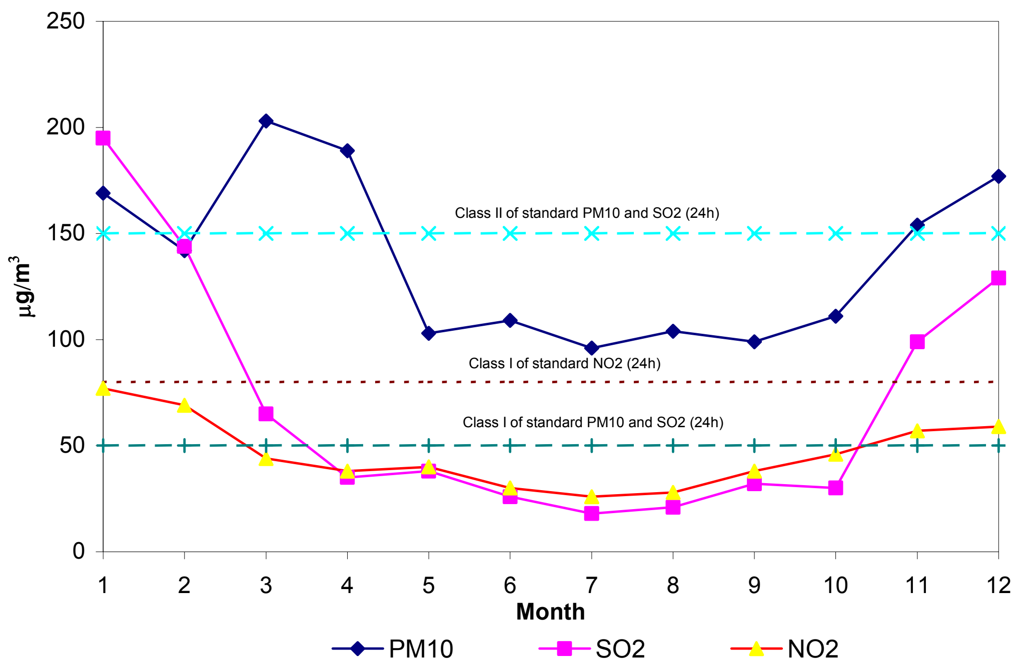


Figure 5.2 Monthly variations of average pollutant concentrations in 2002

5.3 Methodology

A complex mixture of pollutants characterises 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 (Künzli et al., 2000). 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).

5.3.1 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) (see Dockery et al., 1993; Pope et al., 1995; Katsouyanni et al., 1997; Wordley et al., 1997; Prescott et al., 1998; Roemer et al., 1993; and Hitlermann et al., 1998). 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 (Brunekreef and Holgate, 2002). 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 (Xu and Jin, 2003). 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 5.1 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. (1993) and Pope et al. (1995b) in the US. The coefficients are obtained in part from Künzli et al. (2000), a European study which conducted a meta-analysis from indigenous epidemiological studies, and Kan and Chen (2004). The data source and the chosen estimate and CI for 10 μg/m^3 of PM_{10} are shown in Table 5.1.

Table 5.1 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. (1993), Pope et al. (1995b)
Cardiovascular hospital admission	0.013 (0.007, 0.019)	after Künzli et al. (2000)
Respiratory hospital admission	0.013 (0.001, 0.025)	after Künzli et al. (2000)
Chronic bronchitis (adults ≥ 15 yr)	0.045 (0.013, 0.077)	Ma and Hong (1992), Jin et al. (2000)
Bronchitis episodes (children <15 yr)	0.306 (0.135, 0.502)	after Künzli et al. (2000)
Asthma attack (children <15 yr)	0.044 (0.027, 0.062)	after Künzli et al. (2000)
Asthma attack (adults ≥ 15 yr)	0.039 (0.019, 0.059)	after Künzli et al. (2000)
Outpatient visits-internal medicine	0.003 (0.002, 0.005)	Xu et al. (1995)
Outpatient visits-pediatrics	0.004 (0.002, 0.006)	Xu et al. (1995)
RADs (adults ≥ 20 yr)	0.094 (0.079, 0.109)	Ostro et al. (1990)

Aunan and Pan (2004) 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 μg/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. (2000) 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 (Aunan and Pan, 2004).

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. (2000) used 7.5 $\mu g/m^3$ as a reference for PM_{10} as the natural background concentration. Quah and Boon (2003) used both zero concentration and the minimum monthly PM_{10} of 24.7 $\mu g/m^3$ as a reference for Singapore. In a recent study by Kan and Chen (2004), a natural background level of 73.2 $\mu g/m^3$ 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 (2004) and the Tianjin Municipal Bureau of Public Health. We also refer to Xu and Jin (2003) for data for asthma attack and bronchitis episodes. The frequency rate for each health outcome we selected is shown in Table 5.2 in Section 4.

5.3.2 Valuation of the health effects to air pollution

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 criticised 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 (O'Connor et al., 2003). 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 (Mrozek and Taylor, 2002 and Viscusi and Aldy, 2003).

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 (Brouwer, 2000) 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 (Alberini et al., 1996). Some recent valuation studies have begun to address the issue of income and preferences in developing countries. Bowland and Beghin (2001) 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 (2003) 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 (2003) 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. (2001) 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. (2001) 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 (2003) but lies well within the range suggested by Bowland and Beghin (2001) for developing countries. According to this, we derived a WTP of US$85,833 for Tianjin after GDP per capita adjustment. Zhang (2002) also uses the CV method to estimate for Beijing the WTP on mortality in 1999 and derived a unit value range from US$ 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 (2005) 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.

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 (1997) 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.

5.4 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 $\mu g/m^3$ in 2003 was taken for the estimation. The threshold is assumed to be 50 $\mu g/m^3$, which is the national primary ambient standard for PM_{10} in China. Table 5.3 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 5.2 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 5.3 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 (2004) assess the economic cost of air pollution for Shanghai, which equivalent to 1.03% of GDP in 2001. Xu and Jin (2003) 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. (2002) produce a value of 4.3% of GDP for the cost of air pollution in Shijiazhuang in 2000.

Table 5.3 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

Table 5.4 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

As mentioned earlier, the estimation of the cost was conducted based on a threshold of 50 μg/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 5.4 shows the total economic costs if the thresholds are 0 μg/m^3 or 75 μg/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.

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.

5.5 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. (2002) 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 (Tianjin Environmental Protection Bureau, 2004). 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 (Raufer et al., 2002).

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 (2003) 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 (2003), 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. (2005) 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 industrialised 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 (2004) 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.

Chapter 6

Summary and conclusions

Most of China's current environmental problems arise largely as consequences of economic growth or are exacerbated by economic factors. Therefore, it is crucial to take into account economic considerations when dealing with them. However, it is also important to bear in mind that economic factors are only part of the whole image of environmental change. Environmental problems can only be effectively addressed and analysed in a holistic manner involving social and natural systems, as well as the interactions between them through interdisciplinary and/or multidisciplinary approaches. As China's economy has been growing rapidly, the influence of economy on the environment has magnified. The intensified economic activities have brought undesirable environmental change. In turn, environmental change influences the economy and human welfare. Such interdependence has to be examined quantitively and taken into consideration in the formulation of economic policy. Furthermore, economic instruments can be used to manage the exploitation of natural resources and to foster desired environmental change. Therefore, economic analyses of environmental issues has policy relevance not only from a broad economic perspective but also from the point of view of devising economic instruments to address specific environmental problems.

China's environment has degraded in the past two decades. Environmental problems have been addressed to a great extent from natural sciences point of view. Interdisciplinary research that incorporates social, economic, and natural systems is, however, quite recent. This thesis contributes to such research by examining the two major contributors to degrading environment in China: water shortage and air pollution. The different chapters show the relevance of economics for understanding the specific environmental problems and how economics can assist in devising government policies or measures to deal with and alleviate these problems.

Chapter 2 studies the potential application of desalination to deal with water scarcity in China. The main concern behind it is the increasing water shortage throughout China and its constraints on socio-economic development. Desalination, as a rapidly developing technology, draws our attention to analyse its feasibility in applying to China for enhancing water supply. One of the most important factors determining desalination decisions is its cost. However, a few scattered publications on desalination are not sufficient in providing a guideline for the actual cost of desalinated water from different desalting technologies. In view of this information gap, a cost analysis of desalination for two main processes, MSF and RO, is conducted based on existing worldwide desalting plants. Our results show the costs for both processes have declined substantially over time. For MSF, the unit cost decreased from 9 $\$/m^3$ in 1960 to about 0.9 $\$/m^3$ in 2001. For desalting seawater, the cost analysis also indicates that there are economies of scale. As China is a country which lacks experience in seawater desalination, the current estimated cost would be slightly higher than the average world level, perhaps about 1.0 $\$/m^3$ would be an appropriate cost of the MSF process in China at the moment.

For RO, the most popular and the largest share of the world capacity, the average cost of desalination decreased from 4.0 $\$/m^3$ in 1970s to about 0.7 $\$/m^3$ in 2001. The raw water quality plays an important role in the cost of RO. Our results show that the average costs for desalting brackish water are lower than for seawater and wastewater desalting, but higher than for river and pure water. The cost for seawater desalination decreased to a little above 1.0 $\$/m^3$, while that for brackish water reduced to about 0.5 $\$/m^3$ in 2001. Recent lower costs of large SWRO plants indicate that RO has great potential to becoming the cheapest process for seawater desalination. Since most of the RO plants have a relatively smaller capacity than MSF plants and given the fact that the desalination plants with small or medium scales will be the most suitable at the beginning of desalination in China, the RO process is likely to be the first choice. A cost of 0.6 $\$/m^3$ for desalting brackish and wastewater and a cost of 1.0 $\$/m^3$ for desalting seawater by RO would be valid in China.

Based on water demand and supply projections, the water shortage will become more severe over the next 50 years in China. Although taking into account water transferred from the south, water deficiency is estimated to be 16 km^3/yr in 2050 in North China. Some of the water can be potentially provided by application of brackish-, waste- and seawater desalination. Desalination can be particularly convenient for coastal cities. To apply desalination in China, water prices are the major obstacle. Current water prices are still lower compared to the costs of desalination as water is subsidized to different degrees by the government. To eliminate water shortage in the future, water pricing will be an effective economic instrument to conserve water. Water prices should be gradually increased to reflect the cost of supply and the marginal cost of water. Governmental policy should facilitate the water-pricing reforms and step-by-step fill the gaps between the cost of desalted water and actual water prices. In conclusion, desalination can provide reliable water supply and will be ultimately economically feasible, therefore it is advisable to invest in and undertake research on selecting planting and brine disposal sites in the near future. However, planning an actual plant under a specific circumstance needs thorough investigation to accurately assess the final cost that are based on more substantive information and refined data.

Chapter 3 extends the cost analysis in the previous chapter in three ways: firstly, the cost analysis is expanded from two to five main desalting processes; secondly, multiple regression is used to analyse the factors that affect the cost of desalination, and thirdly, the cost of water transport is assessed beyond desalination. The results show that, as with all new technologies, the progress in desalinating water has been rapid. MSF is still the leading process in seawater desalination, followed by the VC and ME processes. Whereas it cost about 9.0 $\$/m^3$ to desalinate seawater around the 1960s, the costs are now around 1.0 $\$/m^3$ for MSF. The RO and ED processes are most often used to treat brackish-, waste- and river water. The unit cost of desalting brackish water has fallen to about 0.6 $\$/m^3$. Due to the lower costs, the expansion of the total capacity of RO plants has been pronounced during the last few years. There is no reason to believe that the trend will not continue in the future. The regressions show that the total installed capacity, the year, the raw water quality and the location of the plant all play a role in determining the unit cost of desalination. The estimations suggest a total installed capacity elasticity of –0.35 for MSF, that is, for every 1% extension of the total installed capacity, unit costs decrease by 0.35%. For the year 2001 alone, the total contracted capacity has increased by about 8%. That would mean a decrease of unit cost by 2.8%. The study also indicates an elasticity of –0.16 for the plant capacity for MSF, implying economies of scale. Similarly, the results show a total installed capacity elasticity of –0.29 for the RO process, -0.40 for ME, -0.26 for VC, and -0.38 for ED. ME and ED learn faster than MSF and RO and may potentially challenge the two dominant technologies; VC is a slow learner and may never be used for anything but niche applications.

In terms of water transport, we found that transporting water horizontally is relatively cheap whilst the main cost lies in lifting it up. We found that desalinated water could be delivered to Bangkok for 1.1 $\$/m^3$, to Phoenix for 1.3 $\$/m^3$ and to Zaragoza for 1.4 $\$/m^3$.

These are probably competitive prices at the moment, and they may well fall in the future. However, getting water to New Delhi would cost 1.9 $\$/m^3$, to Harare 2.0 $\$/m^3$, and to Mexico City 2.4 $\$/m^3$. Desalinated water may be a solution for some water-stress regions, but not for places that are poor, deep in the interior of a continent, or at high elevation. Unfortunately, that includes some of the places with the biggest water problems.

The evaluation could be improved by using more detailed and precise data on operating cost of each plant and be complemented by estimating the cost in detail with a relatively smaller amount but high quality data. Future research could also extend this study to a geographically explicit database, with which the potential application of desalination could be assessed based on information on local water supply in a region.

Chapter 4 focuses on econometric analyses of domestic, industrial and agricultural water uses in China using province level panel data, which provides insights into the responses of consumers to exogenous changes. The examination of water uses shows that the regional disparity in the levels and patterns is considerable. Economically developed or more industrialised areas at the coast consume less water than the agriculture dominated economy in provinces of the west and far south of China. The results suggest that both economic and climatic variables are significantly correlated to water demand. For the domestic sector, income is the dominant factor influencing the magnitude of water use and shows an income elasticity of 0.42. We found that richer provinces have a *higher* income elasticity than do poorer provinces. Availability of water resources and climatic variables also contribute to domestic water use. The coastal area shows a relatively higher consumption than the inland area. For the industrial sector water use per industrial output is negatively related to industrial development, which implies that less water per output is needed when the industry grows and expands. The result suggests an output elasticity of –0.32. Industrial demand is also responsive to price changes and shows a price elasticity of –0.35. As for the agricultural sector, the water use per irrigated land area decreases when agricultural production increases in a region. Rural net income, family size and temperature are positively correlated with agricultural water use.

The estimation can be improved by using panel data covering a longer time period or more disaggregated sub-regional level analyses. It would also be useful to extend the study with more adequate data, especially regarding the time series of water prices for the various sectors. Well-designed household surveys would provide richer information and greater insights into the factors influencing domestic water demand.

Chapter 5 assesses the economic cost of air pollution in Tianjin. Air pollution is associated with respiratory disease and is one of the major causes of premature death. In this study the health impacts from urban air pollution is quantified and monetarily valued for Tianjin. Our results suggest that the cost of air pollution 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 or 3.7% of Tianjin's GDP for the year 2003. Among them, mortality costs are 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 results imply 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. It also underscores the importance of implementation of policies and strategies for mutual development of economy and environment.

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 natural gas, energy consumption is most likely to increase. However, a more efficient energy structure and adopting advanced technology would help reduce pollutants and carbon dioxide emissions considerably. One option is to reduce the coal consumption by gradually replacing it with natural gas. Another option is to use lower sulphur coal to replace coal, which would greatly reduce sulphur pollution. The third option is to replace the small industrial boilers with larger and more efficient facilities that reduce energy consumption and emission in the heating sector. Besides physical measures to reduce emissions, we suggest to impose an effective tax or charge on emission discharge. Without internalising the health costs, natural gas and central heating systems will not be competitive with coal-based small boilers. Although China has employed the Pollution Levy System, 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. Therefore, in order to achieve emission reduction, the levy has to be increased to a reasonable level.

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. Due to the rapid increase of vehicle population the NO_x, CO and ozone problems will increase. In view of the very rapid growth in the vehicle fleet, China's environment could face severe strains with significant public health consequences unless vehicle technology is substantially upgraded and fuel quality improved.

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 can reduce greenhouse gas emissions with financial support through Clean Development Mechanism. Moreover, for China, it has been estimated that the cost of cutting CO_2 emissions by 5-10% would be offset by the benefits resulting from reduced health effects through air pollution (IIASA, 2003). Emission control strategies that simultaneously address air pollutants and GHGs may lead to a more efficient use of the resources on all scales.

In conclusion, China has a long way to go in balancing its natural resources and cleaning up the environment. The outcomes of this thesis demonstrate how economic principles can be applied in the analyses of specific environmental damages, as well as in drawing implications for policy making. The thesis provides useful economic insights into several specific environmental problems, enhances the knowledge of interactions between economy and environment, and of the possible policy instruments to address those problems. It bolsters the significant contribution of interdisciplinary approach in addressing interlinking development problems in China. Although increasing research has been conducted in the field of environmental economics for China, many issues remain to be addressed. In a broad sense, environment should be included in the national accounting system in which the natural resources have values and the degradation of environment counteract the economic welfare (UN, 2003). This is called "Green GDP", which has been under heated discussion in China. In addition, the "polluter pay principle" should be widely disseminated and gradually implemented in environmental regulations (OECD, 1992). The economic policy instruments for management of natural resource usage and environment, such as taxes, charges and tradable property rights, should be strengthened in both designing environmental policies and their implementations (Turner et al., 1993). Moreover, an in-depth economic analysis on to which extent and what scale to apply the water resources fee, pollution discharge fee and emission levies at regional levels should be endorsed by the government in line with the formulation of economic development plan.

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