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Case Study of China

Xiaoliu Yang

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Executive Summary

China's industrial growth has been extremely rapid in the past two decades, with an annual growth rate of about 15% in the 1990s. While this has helped lift tens of millions of people out of poverty, serious environmental deterioration has accompanied this rapid growth.

Prior to 2000, policy instruments in China consisted mainly of taxes and fees and focused mainly on post-pollution management. After 2000, market-based instruments (e.g., discharge permit trading and ecological compensation) attracted more attention but the use of environmental economic instruments is still ad hoc and unsystematic.

Existing policies stress post-pollution management rather than ex-ante reduction and water environment protection. The motivation behind the participation of certain agencies in pursuing environmental economic policies was often self-interest rather than the greater good. This has resulted in a certain level of policy chaos, making environmental economic policies less useful for environmental management and reducing the effectiveness of policy implementation.

China has used environmental economic policy instruments in various forms almost since the start of the initiation of water pollution control policies in the early 1970s, although, even today, they remain supplemental to the command and control system.

Current economic policies to protect water quality include:

- environmental taxes, pollution levies, and wastewater treatment tariffs;
- emissions trading markets (e.g., water pollutant discharge permit trading, watershed water rights trading, and ecological and environmental compensation);
- guaranteed deposits; and
- subsidies and incentives (e.g., subsidies to enterprises for pollution reduction and special subsidies for the construction and operation of municipal WWTPs).

This case study analyses three EPIs in China in depth, namely:

- Pollution charges for industry
- Abstraction charges and Irrigation water pricing
- Phasing out farm input subsidies

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1. Characterisation of China

1.1 Population, Urbanization, and Growth

Population Growth

China has the world's largest population, 1.3 billion people, representing just over one-fifth of the global total. The country's demography has been heavily influenced by government policies over the last 25 years, and the population growth rate is now only 0.6 percent.

Urbanization

The level of urbanization is striking. In 1980, the proportion of urban dwellers constituted less than 20 percent of the population, in 2000 it was 36 percent, and by 2020 it is projected to be 54 percent. The growing urban population has been accommodated through rapid expansion of existing cities and the emergence of new cities; the total number of cities in China increased from 190 in 1978 to 663 in 2000, and have leveled off since then. A critical feature of China's urbanization since the early 1980s has been the expansion at the sub-city level, particularly in categories such as "small cities", "established towns", and "township concentrations".

From 1985 to 2004, the number of established towns increased from less than 8,000 to almost 18,000, accounting for a nonagricultural population of 63.5 million people. Between 1990 and 2004, the established towns category increased by about 14.5 million people, which represents almost a seven-fold increase. By 2004, China's total non-agricultural population below the city level was about 100 million, and about 44 percent of China's total urban (non-agricultural) population was living in small cities, established towns, or township concentrations. In essence, these statistics indicate that the strongest drive for China's rapid urbanization comes primarily from the lower levels below the cities. The increase in urbanization results in a rising demand for water from the established water supply system and an increase in water pollution in the short run.

Economic Development

China is experiencing rapid economic development. GDP grew by 9.4 percent in 2004. Initial information indicates it was 8.3 percent in 2005 and will be about 7.5 percent in 2006. Rapid economic growth has brought about significant improvements in the standard of living for many Chinese, but it is generating increasing levels of demand for water. By relating water demand projections to expected sector growth, projections indicate that this growth will lead to an increase in water demand of 6.5, 32, and 35 percent (2003–2020) from agriculture, industry, and residential users respectively (Chinese Academy for Environmental Planning, 2004). These figures imply that a total increase in demand for water of 83 billion m^3 will be essential if

China is to maintain its current pattern of economic growth. However, with a relatively constant water supply, the increased water demand will have to be met mainly through water savings and improved water quality.

1.2 River Basins

China has nine main river basins: i) Song-Liao River Basin (91 billion m^3/yr); ii) Hai-Luan River (22.8 billion m^3/yr); iii) Huang River Basin (Yellow River) (66.1 billion m^3/yr); iv) Huai River Basin (62.2 billion m^3/yr); v) Changjiang River (Yangtze River) Basin (951.3 billion m^3/yr); vi) Zhujiang River (Pearl River) Basin (333.8 billion m^3/yr); vii) Southwest River Basins; viii) Southeast River Basins; ix) Interior river basins (rivers not discharging into the sea).

Each river basin presents a specific management challenge because of its socioeconomic, climatic, morphological, and hydrological conditions. The most important feature is the abundance of water relative to population, arable land, and local GDP. Table 1-1 provides a summary of water availability, population, and arable land area. Table 1-1 shows the relative water availability in different river basins, of which the northern rivers account for less than 20 percent. It also shows water availability per capita and per hectare, which is as low as 343 and 6,000 m^3 in the north, and as high as 29,427 and 346,350 m^3 in the south. In 2001, the water

Table 1-1. Main River Basins and Major Characteristics

		National Percentage		*Water Availability per capita (to m^3)*			*Water availability per ha (m^3)*
River Basin	*Water availability % (1000 m^3)*	*Population*	*Arable Land*	*1997*	*2010*	*2050*	
Northern Rivers							
Interior R.	4.6 (130.4)	2.1	5.7	4,876	4,140	3,331	23,835
Song-liao	6.9 (192.2)	9.6	20.2	1,646	1,501	1,287	9,900
Hai	1.5 (42.2)	10	11.3	343	311	273	3,885
Huai	3.4 (96.1)	16.2	15.2	487	440	383	6,555
Huang	2.7 (74.4)	8.5	12.9	707	621	526	6,000
North Total	**19.1 (535.3)**	**46.4**	**65.3**	**8,059**	**7,013**	**5,800**	
Southern Rivers							
Yangtze	34.2(961.3)	34.3	23.7	2,289	2,042	1,748	41,745
Pearl	16.7(470.8)	12.1	6.7	3,228	2,813	2,377	67,515
Southeast	9.2(259.2)	5.6	2.5	2,285	2,613	2,231	80,160
Southwest	20.8(585.3)	1.6	1.8	29,427	25,056	20,726	346,350
South Total	**80.9(2277)**	**53.6**	**34.7**	**34,001**	**32,524**	**27,082**	

development ratio (the ratio between water supply and water availability) was 0.2, but the north has a much higher value, up to 0.93 in the Hai River basin. The northern rivers are therefore characterized by high use/availability ratio, water scarcity, and serious pollution problems. Inter-annual and seasonal variations in precipitation are quite large across most of China, but are most pronounced in the water-scarce north. For instance, the difference between minimum and maximum precipitation is generally 3 to 6 times in the northern regions, while in the south it is only 2 to 4 times. In the Hai Luan and Huai rivers, the flow is less than 70 percent of the average once in four years and less than 50 percent once in 20 years. However, the northern rivers are also highly regulated: the annual storage capacity in the 3-H basins is about 90 percent of the average annual runoff (the country wide average is about 20 percent.

2. EPI Background

2.1 Background

2.1.1 Water Availability

China's total annual renewable water resources amount to between 2,400 and 2,800 billion m^3/year (6^{th} in the world). However, annual per capita water availability was only 1,856 m^3 in 2004 (average 2,100 m^3 2000 - 2004), which is about a quarter of the world average (8,513 m^3/year). The south is relatively water abundant.

Water scarcity is very severe in northern areas, where average annual per capita availability is only about 725 m^3. However, population growth will continue to undermine per capita water availability.

When China's (projected) population peak occurs, at around 1.6 billion in 2030, annual per capita water availability will be only 1,750 m^3. Given that China cannot increase its water resource base, future demand can only be met by increasing water efficiency in municipal, industrial, and/or agricultural sectors, promoting water re-use, or by cleaning up water that is currently unfit for consumption.

Precipitation patterns across the country show that the rainy season is as long as six to seven months in some southern areas and as short as two or three months in more arid northern regions. In general, annual precipitation decreases from the southeast to the northwest. In eastern areas around the Changbai Mountains, annual precipitation may reach 800 to 1,000 mm - about 800 to 900 mm in the area from Qinling Mountains to the Huaihe River (Anhui province), above 1,000 mm south of the middle and lower reaches of the Yangtze River, and more than 2,000 mm in some coastal mountainous and hilly areas in the southeast and parts of the southwest. In the western regions (except for the Altay and Tianshan Mountains), most areas are dry. In the Tarim and Qaidam basins, annual precipitation is less than 25 mm; mean annual precipitation at some stations in the Turpan basin is less than 10 mm.

2.1.2 Water Use

The countrywide average use/availability ratio is about 20 percent, which, seen in isolation, is not an alarming value, but at local levels there are many areas where water systems are stretched beyond their capacity. In many areas, the cost of water shortages from pollution appears severe. According to 1997 statistics for the Yellow, Huai, and Hai-Luan river basins, exploitation rates reached 67, 59, and 90 percent respectively (as compared to use rates of below 20 percent in the south).

With about 40 percent utilization rate, all of these northern rivers exceed international recommendations for water use. Moreover, usage in the Hai-Luan basin exceeds the sustainable yield, resulting in groundwater depletion. The net effect is that the bulk of river flows in the north comprise wastewater, and the dilution and absorptive capacity of the rivers is severely compromised.

The total amounts of water use and wastewater generation have actually declined in recent years (World Bank, 2005; China Statistical Yearbook, various years). It is revealed that total water consumption declined by 4.5 percent between 1997 and 2005, and the relative importance of agricultural, industrial, and consumption categories respectively accounted for 63.3, 22.8, and 12.5 percent of total water usage in 2005. The overall decline is due to reduced demand from the agricultural sector and has occurred in spite of increases in industrial and consumption usage.

Water savings from agriculture have been achieved through investing in more efficient irrigation systems and cultivation methods. In rice production, for example, there has been a widespread shift from traditional to water-saving irrigation systems that reduce water consumption by a third (FAO, 1996).

These efficiency gains have allowed the overall water demand from irrigation to fall by about 5 percent (1997-2005), while the total irrigated area actually increased by about 5.5 percent over the same period. The declining proportion of water usage accounted for by agriculture is even more impressive when we consider that in 1980 it accounted for around 80 percent of total water use, and it is projected to be close to 50 percent by 2050 (World Bank, 2001). The growth that has occurred in the consumption category is driven by urbanization (4.3 percent), and rising urban per capita residential consumption (110 l/cap/y 1980 - 230 l/cap/y 1997). Urban and rural residential consumption both account for around 5.5 percent of water consumption. In 2001, urban water consumption surpassed rural consumption.

For a long time, China experienced rapidly increasing per capita urban water consumption rates; in recent years, this trend seems to have leveled off. The proportion of people with access to an improved water source (i.e. household connection or public standpipe) in rural and urban areas is 66 and 94 percent respectively in 2002 (WHO, 2004). However, other estimates are less optimistic and indicate that as much as half of the population does not have access to clean water.

Of particular concern are the communities at the town and township level (i.e. below city level), which may not be captured in typical urban-rural statistics. China has 18,000–20,000 town and township centers, often with very poor access to clean water, but because of their hybrid (urban–rural) status, the Chinese institutions that would normally be responsible for infrastructure development have no clear mandate.

2.1.3 Drinking Water

Water supply in China has increased significantly in both urban and rural areas in the past fifteen years. According to the Chinese Ministry of Health (MoH) statistics, the percentage of rural population with some kind of access to a drinking water supply rose from 75.5 percent in 1990 to about 93.8 percent in 2004, an increase of about 220 million beneficiaries. In urban areas, access to tapped water rose from 48 percent in 1990 to 88.9 percent in 2004. While these increases in the supply of drinking water represent a major achievement, trends in the quality of water supply are less encouraging.

According to a Ministry of Water Resources (MWR) survey, more than 300 million rural residents throughout the country consume unsafe drinking water. In Chongqing municipality, where the World Bank conducted a study at both town/township and pure rural areas to examine the correlation between access to clean water and public health, 39.8 percent of the population have no access to a safe water supply. The health risks associated with both, biological or microbial pollutants (e.g. large intestine bacilli, hepatitis B virus, cholera virus, typhoid, E-coli etc.) and chemical pollutants (e.g. heavy metals, fluorine, arsenic, benzene, oil, etc) are widespread. According to MWR, an estimated 63 million rural people in the northern, northwestern and north-eastern provinces and across the Huang-Huai-Hai (3-H) plains are exposed to drinking water with high fluorine content. In the coastal areas of North and East China, salinization of drinking water sources is affecting about 38 million people, and some 2 million people in parts of Inner Mongolia, Shanxi, Xinjiang, Ningxia and Jilin drink water with high arsenic content, which has been linked to several types of cancers.

Disease incidence and mortality rates due to microbial pollutants remain relatively low on national level. For example, according to the China National Health Survey, in 2003, infectious diarrhea (ex. bacterial and amoebic dysentery) and typhoid incidence were 35 and 4.17 cases per 10,000 persons respectively.

However, the incidence rates in towns, townships and villages, particularly in heavily polluted areas, are suspected to be significantly higher. Increasing the supply of clean drinking water, especially in rural areas, has become one of the major objectives of the Government of China (GoC).

The GoC began addressing the issue through an ambitious $2.1-billion rural drinking water supply project, completed during the 10th Five-Year Plan (2001–2005). The project included the installation of 800,000 new water processing facilities, which provided access to clean drinking water to 14 million rural households.

According to the MWR, the overall project is estimated to have relieved water shortages for more than 57 million rural residents. The continued problems associated with the lack of access to clean water, have prompted the Chinese government to continue with its aggressive measures to tackle the issue. In urban areas, stricter standards for piped water quality have taken effect since June 1, 2005. In rural areas, according to MWR, China plans to cut down the number of residents without access to clean drinking water by one third by 2010 and to provide safe drinking water to all rural residents by 2020.

2.1.4 Urban Water Supply

The rapid growth of China's urban centers has necessitated increased level of water infrastructure. In 2003 China's national urban water supply capacity was 87.5 billion m^3/year. This represents a vast increase in water capacity in China over the last 25 years. The increase in urban population supplied with water services grew rapidly from 62 million in 1978 to 291 million in 2003. The urban water supply capacity increased at a similar rate, by almost an order of magnitude in the same 25-year period. In order to achieve this capacity increase there has been a nine-fold increase in the length of China's water supply pipelines, from 35,986 km in 1978 to 333,289 km in 2003. However, since the late 1980s total urban supply capacity has greatly exceeded the water actually supplied to the consumers. In 2003, daily capacity was 239 million m^3, but only 54 percent (130 million m^3) was supplied.

An important element is the leveling-off that has occurred in total water supply since 1994, in spite of continued increase in residential use. This pattern is thought to be related to reductions in industrial water demand connected to government initiatives to address water use and pollution from this sector. However, it also shows how urbanization is quickly negating the gains made in mitigating industrial water use.

2.1.5 Water Quality

China has established a water quality classification system based on purpose of use and protection target, following Environmental Quality Standard GB3838-2002:

- Grade I – Mainly applicable to the source of water bodies and national nature preserves.
- Grade II – Mainly applicable to class A water source protection area for centralized drinking water supply, sanctuaries for rare species of fish, and spawning grounds for fish and shrimps.
- Grade III – Mainly applicable to class B water source protection area for centralized drinking water supply, sanctuaries for common species of fish, and swimming zones.
- Grade IV – Mainly applicable to water bodies for general industrial water supply and recreational waters in which there is not direct human contact

with the water.

- Grade V – Mainly applicable to water bodies for agricultural water supply and for general landscape requirements.
- Grade V+ – Essentially useless.

Based upon this classification, water quality is being monitored on regularly basis in almost 500 monitored stations within China's referred nine main rivers basins through national and provincial-run water monitoring centers. The number of monitored sections within each river basins varies from for example 104 in the Yangze basin to only 17 in the South West rivers. The most polluted water sections are largely in Northern China and particularly in the most populous provinces of Henan, Anhui, Jiangsu, Hebei, Beijing and Tianjin. Other water pollution hotspot areas includes North East provinces and in high population concentration in Sichuan and Chongqing.

Results from ongoing water quality monitoring show overall water quality in China to be poor. In 2004, only 28 percent of monitored river water was in grades I to III, while as much as 31 percent was in the worst two categories. The situation does not appear to have improved and larger shares of the water quality appears to have increased between 2000 and 2004 in particularly the northern regions. Among the about 30 pollutants included in the overall water quality monitoring schemes, usually about 14 are selected in comprehensive water pollution indexing. The worst individual monitored pollutant establishes the water quality grade for the section. The most important indicator for triggering water quality levels was nitrogen (in the form of ammonia), followed by organic materials (BOD and COD). The relative importance of different indicators in terms of the frequency with which they 'trigger' a lower water quality category in China's main rivers. Many rivers have a similar pollution structure, but the Huai is heavily dominated by nitrogen (as ammonia), whereas in the Yangtze, nitrogen is never the most important pollutant.

China's large lakes are also experiencing declining water quality caused by both point and non-point source emissions. Lake Dianchi is a good example of this trend, as water quality declined from Grade II in the 1960s, to grade V and IV in the 1990s. The lake has also undergone significant eutrophication during this period with a massive shift from a high biodiversity low productivity system to a low biodiversity high productivity state. Concentrations of organic matter and nutrients, such as phosphorus and nitrogen, show high levels, and the latter are still increasing. Organic material contributes to decreased oxygen levels and bacterial growth and nutrients cause eutrophication. Pollution sources are often grouped in two classes:

- Point source pollution is made up of industrial and municipal emissions. Recent measures to encourage industries to meet wastewater regulation standards led to a 25 percent reduction in emissions (28 to 21 billion tons) between 1990 and 2004. In 2000, industrial sources accounted for 11 percent of BOD, 4 percent of Total Nitrogen (TN), and 2 percent of Total Phosphorus

(TP) discharges. Municipal sources are increasing, as population and economic growth leads to more wastewater, important elements include growth in flushing toilets and washing machines. Municipal sources accounted for 52 percent of BOD, 69 percent of TN, and 53 percent of TP in 2000.

- Non-point sources are primarily related to agricultural activities, including fertilizer and pesticide run-off from farmland, and infiltration of livestock waste. In 2000, non-point sources accounted for 37 percent of BOD, 27 percent of TN and 45 percent of TP. The excessive loads of nutrients, and in particular Phosphorus, leads to eutrophication, excessive algae growth, reduced biodiversity levels, and poor quality water.

Despite considerable efforts to clean up China's major river basins the situation remains generally poor. There have been some improvements in the Yangtze and Pearl River basins where a reasonable proportion of the water is now classified at grade I or II (but all still contain areas of very poor water quality, particularly in the tributaries). However, many of China's rivers, such as Hai, Liao, Yellow and Songhua, are still dominated by water of the worst categories (V and V^+). The problem is typically most prominent when rivers flow through large cities, where discharges of organic materials have caused increased concentrations levels of various pollutants.

The clean-up challenge is huge and is especially important given the water shortage problem. According to WB estimates, the cost of water shortages from pollution ranges from 1 to 3 percent of local GDP in water scarce areas (World Bank 2006). In order to address this problem, the Chinese government has already started work on a number of water transfer projects (south to north), but poor water quality in intervening rivers is a major constraint.

Addressing water pollution in China is also significant given its particularly high health cost and to some extent agricultural costs. The estimated cost in 2003 for the whole of China for water related damages of four major types of crops (wheat, corn, rice and vegetable) was about 0.05 percent of GDP. Establishing the true extent of public health impacts from water pollution is challenging, because it is hard to isolate specific dose response functions given the wide range of factors, including food chain effects. Initial World Bank analysis on the environmental health impact of water pollution in China found significantly higher disease rates, e.g. cancers and spontaneous abortions, among fishing and farming communities living near polluted water sources (World Bank, 1997).

Ongoing work has indicated that improved water quality could significantly reduce the spread of the hepatitis A virus (70 to 90 percent of hepatitis A cases in China are transmitted by water). Waterborne diseases, such as diarrhea, cholera, and typhoid, which are entirely related to impure water, could be reduced by almost 50 percent by moving from heavily to moderately polluted water (World Bank, 2006). The ongoing study also estimates 9 million cases of diarrhea due to water pollution based upon the national health survey from 2003. Excessive application of fertilizers can also have health consequences; under aerobic conditions, NO_3 can be formed, which is absorbed into the body and interferes with the blood's oxygen carrying capacity.

Average water quality for China as a whole shows a steady increase in the relative abundance of the worst and best water quality categories. Between 1991 and 2005 the number of monitoring stations recording the worst categories (V and V^+) stayed the same at about to 30 to 35 percent (but with high fluctuations), while the percent of best grades (I and II) increased from 3 to 20 percent. This may reflect two main processes: (a) the difference in water quality patterns in the north and south of the country; and (b) water quality improvements in the main course versus deterioration in tributary waters.

The trend between recent water quality in southern rivers (Yangtze, Pearl) and northern rivers (Hai, Huai, Yellow, Liao, and Song) is different. In the southern rivers, the proportion of water rated in the best categories has increased from 2 to almost 60 percent, while in the northern rivers the change is much smaller (4 to < 10 percent). However, water in the worst categories has increased in northern rivers, from 40 to 45 percent (but with fluctuations up to > 60 percent), as a result of the pollution sources outlined above and the very low levels of water available to absorb it. The deterioration of water quality is particularly severe in the Hai and Huai river basins.

Another important reason for the simultaneous increase in abundance of best and worst categories is the fact that much of the attention in cleaning up operations has been focused on the main river course, with less effort on the tributaries. For example, in the Yangtze River, which has experienced significant water quality improvements in recent years, there is a stark difference in the relative abundance of best and worst water qualities between main river and tributaries.

In 2001, the main river had no cases of class V or V^+, while this accounted for 48 percent of the water in its tributaries. An important indicator of the pollution management efforts that have been made over the last decade is the decline in chemical oxygen demand COD) from industrial sources in many rivers. Improvements can be found in the Liao, Hai and Huai rivers.

3. General Assessment on EPIs

China has established a comprehensive water environmental-economic policy system that entails discharge levy, sewage treatment fee, discharge permits trading, PPP, and public finance. However, there remain problems and gaps which need to be addressed to improve the effectiveness of the system.

3.1 Existing EPI System

China has used environmental economic policy instruments in various forms almost since the start of the initiation of water pollution control policies in the early 1970s, although, even today, they remain supplemental to the command and control system. Current economic policies to protect water quality include:

- environmental taxes, pollution levies, and wastewater treatment tariffs;
- emissions trading markets (e.g., water pollutant discharge permit trading, watershed water rights trading, and ecological and environmental compensation);
- guaranteed deposits; and
- subsidies and incentives (e.g., subsidies to enterprises for pollution reduction and special subsidies for the construction and operation of municipal WWTPs).

3.2 Motivation

Prior to 2000, such policy instruments in China consisted mainly of taxes and fees and focused mainly on post-pollution management. After 2000, market-based instruments (e.g., discharge permit trading and ecological compensation) attracted more attention but the use of environmental economic instruments is still ad hoc and unsystematic.

Existing policies stress post-pollution management rather than ex-ante reduction and water environment protection. The motivation behind the participation of certain agencies in pursuing environmental economic policies was often selfinterest rather than the greater good. This has resulted in a certain level of policy chaos, making environmental economic policies less useful for environmental management and reducing the effectiveness of policy implementation.

3.3 Assessment

The environmental economic instruments are classified into three main categories:

3.3.1 Fee-Based Instruments

The Pollution Discharge Levy The pollution discharge levy system is China's most comprehensive environmental economic instrument due to its wide coverage and significant effect. The levy encourages enterprises to reduce pollution and provides a source of revenue for the construction of pollution control facilities for enterprises, pollution control for key pollution sources, and the operation of environmental management agencies. The total volume of collected revenues for the country as a whole has grown annually, from CNY1.2 billion in 1986 to CNY17.8 billion in 2007. A total of 647,335 enterprises are covered by the levy system. Research in 1997 showed that pollution levies were more effective than other pollution control measures in stimulating enterprises to adopt environment-friendly technologies. Nevertheless, there remain many issues and problems:

- Low levies and inadequate management. The cost of the levy is lower than the cost of pollution reduction so the deterrence value is low. Fines for violations are often set too low, so that it is often cheaper for the polluters to pay the fine than to solve the problem. Administration of the levies is somewhat lax in some regions (e.g., failure to collect levies on a regular basis, failure to adjust the levies for many years) due to lack of enforcement capacity and/or lack of political will to introduce unpopular levy adjustments.
- Non-collection. Environmental authorities at various levels focus the pollution collection effort on industrial enterprises but neglect animal and poultry breeding and the catering industry. They also tend to focus on large enterprises and pay less attention to numerous highly polluting small enterprises. Inventories of pollution sources often include false reporting and under-reporting.
- Low collection capacity. Many environmental supervision agencies responsible for levy collection are under-staffed, severely affecting levy collection and other law enforcement activities.

Wastewater Treatment Fee

China began collecting wastewater treatment fees in 2002. The rates are determined by the concerned local government. Wastewater treatment fees are normally incorporated into the water tariff and collected through the water bill. The revenues collected are either submitted to the water board or the finance bureau of the local government before forwarding to the WWTPs as payment for the sewage treatment service rendered. In 2007, wastewater treatment fees were introduced and collected in all the provincial capital cities, autonomous regional capital cities (except Lhasa), and centrally administered municipalities.

The total collection amounted to CNY11.0 billion with an average rate of CNY0.77 per cubic meter (m3). By the end of 2008, most cities collected wastewater treatment fees and a total of 1,521 municipal WWTPs were in operation in China treating 66.8 million cubic meters per day (m^3/d) of wastewater, against the design capacity of 90.9 million m^3/d.

There is considerable potential to improve collection of wastewater treatment fees. Existing problems include vast regional differences in fee levels and fee collection efficiency, improper coverage, low rates, collection difficulties, lack of flexibility with WWTP operation, heavy command-and-control fee collection methods, and inappropriate measurement.

In general, the fee rates do not cover the cost of operation of most WWTPs so that they are not self-financing. Most WWTPs are over-staffed and inefficient. The tariff system is inequitable insofar as the fees are collected based on the quantity of effluent discharge as a portion to water consumption.

The current tariff system does not take into account the composition and severity of the pollutant discharged, i.e., the actual treatment service provided. The regulation of the performance of WWTPs is lax.

3.3.2 Allowance-Based Instruments

Discharge Permit Trading

Experimentation with discharge permit trading began in the late 1980s (water discharge permit trading in Shanghai) but it was not until 1996 that the State Council approved a national major pollutant discharge control program, which initiated nationwide implementation of the discharge permit system—the basis for discharge permit trading for water.

Further experimentation with discharge permit trading was undertaken during the early stages of the 10th FYP although it was very localized. In 2007, the Ministry of Finance (MOF) and Ministry of Environmental Protection MEP approved an experiment for pay-based allocation of discharge permits and trading in the Taihu Lake catchment in Jiangsu Province.

Since then, an increasing number of commercial enterprises have joined the field of discharge permit trading. Traders are varied and even include environmental organizations and interest groups.

In March 2008, the Wuhan Guanggu Property Exchange established a discharge permits trading platform. The Beijing Environment Exchange, the Shanghai Environment and Energy Exchange, and the Tianjin Discharge Permit Trading Exchange have been established to trade in a wide range of instruments including SO_2, COD, and other traditional pollutants as well as greenhouse gas emission rights, patents, and technology copyrights.

This emerging interest represents dramatic progress for discharge permit trading, but there remain many challenges to be overcome before the system achieves its true potential. Major challenges include the following:

- Inadequate legal framework and operational guidelines. There are no national laws to define discharge permits, rules on how to trade permits, protect the interests of trading parties, resolve disputes, etc.
- Weak discharge measurement, monitoring, and supervision. Accurate measurement and strict monitoring of pollution discharges through strong supervision are critical to properly functioning discharge permits trading but the current system is weak due, amongst other things, to a general lack of automatic monitoring equipment in relevant enterprises.
- Lack of independent discharge permit trading market. So far, all pollution trading has been conducted under the coordination of local environmental

protection bureaus (EPBs). The trading market between permit holders is yet to be developed. Permit holders tend to hoard their permits obtained from EPBs due to their perception of an increasingly strict pollution reduction requirement.

- Poor pricing mechanism. Prices tend to be actually set administratively by the local governments rather than through a market-based pricing mechanism. Such a pricing mechanism may increase, among others, the rent-seeking behavior of the concerned governments at local level.

3.3.3 Input-Based Instruments

Public Finance Policy

China's annual investment in environmental protection currently amounts to about 1.5% of GDP. There are basically three sources of funding for environmental protection: public finance, bank finance, and enterprise and private finance.

During the 10th FYP, public financing for environmental protection accounted for 13% of the total, while 15% came from bank financing, and 72% came from enterprise and private financing. The main sources of public financing are the "211 environment protection account" in the government budget, the special environmental protection funds, environment and ecological compensation payments, and environment-friendly tax policies.

The 211 Environment Protection Account

As part of the MOF's reform program for government revenues and expenditures initiated in 2006, a specific budgetary account—the 211 Budgetary Category for Environment Protection—was established to track environmental expenditures which, prior to that time, had been scattered across a wide range of programs.

This 211 budgetary category includes expenditures for environmental protection, management, and services; environmental monitoring; pollution control; ecological protection; natural forest protection; reforestation, desert sand control and pasture management; and grassland restoration.

Surveys undertaken under the study show that the 211 budget category has not been mainstreamed down to all levels of government, particularly in the municipalities and counties. The 211 environment protection account further shows that allocations for environment protection have generally not kept pace with general increases in fiscal outlays since its establishment. Environmental expenditures in 2007 were 14.5% higher than they were in 2006 while the overall fiscal outlays increased by 23% over the same period.

Special Environmental Protection Funds (SEPFs) SEPFs are financed by revenue collected from pollution discharge levies and they are the main source of funding for ecological compensation in the China. SEPFs are established for specific purposes and for fixed time periods (e.g., five years) and are dispersed to underwrite projects identified through a bottom–up selection process.

SEPFs are the central government's public financing source for environmental protection, including the central environmental protection special fund, special funds for major pollutant emission reductions, the special funds for water pollution control under the "three rivers and three lakes" program and the Songhua River rehabilitation program, the special fund to provide financial incentive to encourage construction of the urban WWTPs, the special fund for environmental law enforcement in the western region, special funds for nature reserves, and the special fund for rural non-point pollution control.

A total of CNY27.7 billion was allocated to the central government's SEPFs in 2008. SEPFs provide strong financial support for environmental protection, but for now they are mainly special environmental fund for contingencies. As they lack long-term and effective integration, their impact is not obvious. The government is planning to adopt a new financing mechanism to disburse funds from SEPFs upon project completion. This new financing mechanism is intended to encourage a resultsbased environmental management approach.

State Bonds

The central government issues bonds to finance environmental protection. This has become an important new funding source for environmental protection. Between 1998 and 2005, the central government allocated CNY61.6 billion, funded through the state bonds to support the construction and upgrading of the municipal WWTPs across the country.

Fiscal Transfers and Ecological Compensation

Following the 1994 tax reform, the central government increased its share in total fiscal revenues, and fiscal transfer has become an important means for balancing regional and local developments across the country. Since 1998, the central government has increased its budgetary allocation for ecological protection and the scale of fiscal transfer for ecological rehabilitation and natural forest protection.

At present, the central government is actively implementing a pilot ecological compensation policy. As an example, in 2008, the central government made fiscal transfer payment of CNY14.8 billion to the local governments involved in the south-to-north water diversion project through the ecological compensation funds.

Public finance through fiscal arrangement, as it relates to environmental investment, has the following shortcomings in China:

- Unclear responsibilities. The government continues to play a major role in making investment and operation decisions. The 'polluter pays' principle has no sound institutional basis and negative externalities of pollution have not been fully internalized.

- Irrational allocation of fiscal resources and government responsibility. The existing public finance system creates inherent conflicts between fiscal resources and environmental responsibilities among the central and local governments. The dividends and profits of the national state-owned enterprises are centralized at the national government, while pollution controlling duties are left to the local governments. In the poverty-stricken and underdeveloped areas, the local governments always find it difficult to make investment for pollution control and to undertake their environmental management responsibility.
- Lack of integrated environmental management plans to ensure coherent investment. A large part of the recent increases in government environmental expenditures has been programmatic or in response to emergencies. There is a lack of continuity and predictability of budget resources earmarked for most environmental expenditures. There is too much variability in environmental enforcement capacity among different levels of the government.

Public–Private Partnerships

Public–private partnership (PPP) is a cooperative model between the public sector and private enterprise. PPP, for the government, has the attraction of potentially reducing demands on the public budget and increasing the efficiency and effectiveness of service provision. For the private sector, PPP provides access to long-term, reliable, and potentially profitable cash flows.

For China, the attraction of PPPs for WWTP development increased significantly as investment in WWTPs started ramping up significantly under the 10th FYP. By May 2008, 1,408 WWTPs had been constructed in the PRC, of which 448 (about 32%) had been developed and/or operated on a PPP model.

Further development of the PPP approach is being hampered by a number of systemic shortcomings including:

- Low tariffs and insufficient guarantees regarding revenue streams. On average,
- the treatment fees collected equal to only about 67% of actual operational costs. This provides little attraction for the private sector. The private sector also has concerns as to the willingness of local governments to raise tariffs in line with rising costs.
- Concerns about weak government supervision. Too much private sector participation in the provision of an essential public service might be seen as representing a loss of control.
- Lack of supporting policies and operation guidelines. The existing market and franchise policies only provide a guidance framework and lack an appropriate legal basis for PPP arrangements. There are questions as to whether the local governments have the legal right to enter into such

agreements. Some build–operate– transfer agreements have been signed by government-affiliated investment companies, which again raises the question of legality.

3.3.4 Laws, Regulations, and Institutional Arrangement

Over a period of several decades, the PRC has developed a comprehensive range of laws and regulations which provides a sound basis for dealing with water pollution problems. Nevertheless, there remain a number of persistent problems.

The Law on Water Resources and the Water Pollution Prevention and Control Law are the basic laws governing water pollution management in China. In accordance with these laws and related regulations, the responsibility for water pollution control have been shared among a number of ministries, including the MEP (responsible for overall water environmental protection, supervision, and management), Ministry of Water Resources (MWR, responsible for overall water resources management), and Ministry of Housing and Urban/Rural Development (responsible for coordinating the construction and operation of urban sewage treatment facilities).

This is a quite complicated institutional setup that has great potential for improvement. For instance, water pollution needs to be controlled through integrated intervention guided by a coherent strategy.

It is difficult, however, to make a clear division between the two functions of water resources management and water pollution control. The Law on Water Resources seems to specify that water environmental protection plan (i.e., responsibility of MEP) is part of the water resources management plan (i.e., responsibility of MWR). This seems to create an overlapping of the mandates of the two ministries, as the MWR tends to claim an oversight role on all water-related concerns including water pollution management, while MEP has its own system to formulate water pollution prevention and control plans and programs in accordance with the Water Pollution Prevention and Control Law.

The situation is further complicated as the local governments are charged with the responsibility to deliver the local environmental management function and achieve the water pollution control targets specified for their territories in accordance with the policies and regulations set by different ministries, which are often issued in accordance with the respective mandates of specific ministries with little inter-ministerial consultation and coordination.

4. Pollution Charges for Industry

4.1 Policy Context

China's industrial growth has been extremely rapid in the past two decades, with an annual growth rate of about 15% in the 1990s. While this has helped lift tens of millions of people out of poverty, serious environmental deterioration has accompanied this rapid growth. Many cities in China have been among the worst polluted urban areas in the world.

China has adopted various policy measures to control industrial pollution5, which include command-and-control approaches, administrative measures, economic instruments, as well as public disclosure. New sources are subject to an environmental impact assessment policy and a 'three simultaneous steps' policy, which requires pollution abatement facilities be designed, installed and operated simultaneously with industrial production process technologies. For older sources, pollution discharge standards have been designed and implemented for different industries, different pollutants and different areas. Air, water and land have been classified into different zones according to different environmental sensitivities, with different ambient and discharge standards. According to the importance of particular pollution sources, industrial firms are also classified into four government supervision categories corresponding to national, provincial, municipal and county-level government in order of descending importance with the most important being classified directly under the national authorities'monitoring and enforcement activities. Different levels of government can issue penalties or even shut-down orders depending on their corresponding pollution sources.

The pollution charge has been one of the most important pillars of the industrial pollution regulatory system in China. This policy instrument was originally designed to promote compliance with the pollution discharge standards. The Chinese environmental protection law specifies that 'in cases where the discharge of pollutants exceeds the limit set by the state, a compensation fee shall be charged according to the quantities and concentration of the pollutants released.' In 1982, after 3 years of experimentation, China's State Council began a nationwide implementation of the pollution charge. Since then, billions of yuan (US$1=6.3 yuan) have been collected each year from hundreds of thousands of industrial polluters

for air, water, solid waste, and noise pollution. For example, in 1996, the system had been implemented in nearly all counties and cities, and collected over four billion yuans from about half a million industrial firms. The number of firms participating in the system and the collected amount have been rising year after year.

There are some unique features associated with the charge system. For wastewater, this system only imposes charges on the pollutants over the standard, among which only the pollutant violating the standard to the greatest degree enters into the calculation of the total levy fee. In other words, fees are calculated for each pollutant in a discharge stream and the polluter need only pay the amount with the highest value among all the pollutants. The Chinese central government constructs a uniform fee schedule; however, the implementation in different regions is not uniform.7

The levy collected is used to finance environmental environmental institutional development and administration and environmental projects, and to subsidize or to loan to firm-level pollution control projects. A firm can receive up to 80% of the levy paid as a subsidy or loan to finance its pollution abatement projects. To make the levy collection system effective, a schedule of penalties is also specified.

Penalties are collected and utilized by local environmental authorities. Although studies have been conducted to reform the levy system, with most analysts recommending raising China's pollution charge rate, few empirical analyses have actually investigated the polluters' response to the existing charges. In Wang and Wheeler (1996), province-level data on water pollution was analyzed and it was determined that China's levy system had been working much better than previously thought. The results suggest that province-level pollution discharge intensities have been highly responsive to provincial levy variations. This analysis furthers this effort by estimating the responsiveness of a firm's pollution abatement efforts to the structural policy environment.

4.2 Implication

Only a few empirical studies have been conducted in analyzing a firm's behavior in complying with pollution regulations. Previous analyses mostly focused on a firm's pollution discharge, usually associated with external pressures such as the strength of regulation and social norms as well as with a firm's internal characteristics.

There are few empirical studies on pollution abatement investment efforts, with most of them being associated with the impact of environmental regulations on industrial productivity. Smith and Sims (1985) performed an econometric analysis of the impact of pollution charges on productivity in the Canadian brewing industry and found that pollution charges have a negative impact on productivity growth. Gray and Shadbegian (1995) analyzed the relationship between productivity and pollution abatement expenditures for plants in the paper, oil and steel industries, and found that plants with higher abatement cost had lower productivity.

In contrast, this study analyzes an industrial firm's investment and operation expenditures on end-of-pipe wastewater treatment in China, and looks at the determinants of industrial pollution abatement efforts. The pollution charge has been found to have a significant and positive impact on both investment in pollution abatement and operation expenditures of treatment facilities. The elasticities are 27 and 65%, respectively.

While the levy rate has been generally regarded as low in China, the levy could affect abatement expenditures in two ways. One is that the levy itself does provide a strong incentive for firms to invest in pollution abatement so that a lesser charge is paid. Another channel is that higher levies could generate higher possible subsidies which firms can use to invest in pollution abatement. The current data set is not sufficient for a test of possible separate effects.

Studies on the pollution charge are consistent with findings by Wang and Wheeler (1996) where a provincial-level panel data set was used. In yet another study by Dasgupta et al. (2001), inspections, rather than the pollution charge, are found to be the dominant factor in explaining a firm's environmental compliance.

However, the data set used by Dasgupta et al. (2001) is for only one city (Zhenjiang), where presumably the variance of the pollution charge would not be as large as with a nationwide sample as employed in this study.

Overall aggregated policy enforcement effort, approximated by environmental staffing per firm, has a positive impact but it is not significant in either model. Regional enforcement effort should be one of the determinants of the pollution charge. When the pollution charge variable is included in the models, the results show that the enforcement effort variable has no significant, independent effect on pollution abatement.

The environmental zoning policy variable was found to have no significant impact on the abatement decision of existing plants included in the sample. The national government's supervision effort also made no difference in plant-level pollution abatement. However, since the sample only includes top polluters in China, one should be cautious in extrapolating the results to all scales of polluting activity.

In conclusion, the China pollution charge system has provided a strong, positive incentive for abatement expenditure of large and medium sized industrial polluters, and therefore on overall pollution reduction. Increasing the pollution charge rate could be an effective way to further reduce industrial pollution discharges in China. As to how high the pollution charge rate should be, this would be deserving of further research into deriving an optimal pollution charge.

The pollution levy system (PLS) applies only to industrial sources and covers water discharges as well as air emissions, solid waste, noise and radioactive substances. Sources such as municipalities, hospitals and schools are exempt. From the outset, the PLS, initiated in 1978, was viewed as a means of implementing the polluter-pays principle and providing a (major) source of funding for provincial and local Environmental Protection Bureaus (NCEE, 2004). A key feature of the PLS is that 80% of the funds collected are returned to the enterprises for pollution control investments. Initially imposed on discharges exceeding the effluent standard, since 1993 the pollution charge has been extended to all discharges. The level of the charge was initially based on pollutant concentrations at the point of release, rather than mass or volume. In 1993 volume became a determinant.

While the PLS seems to have been reasonably effective in reducing pollution, other factors such as responsibility contracts signed by enterprise managers and local government officials as part of the five-year planning process may have been more important in determining the pollution intensity of industrial activity. The charge rate increase has been much lower than incremental pollution control costs, reducing its influence on polluting behaviour. As a result, the proportion of total charge revenue to the value of industrial output has decreased. The coverage of the charge is another issue. Many township and village enterprises are not levied because local Environmental Protection Bureaus do not have the resources to pursue all sources within their jurisdiction or find that the potential revenues from levying smaller sources do not justify the effort. Since such enterprises generally use less advanced technologies, one would expect them to be paying relatively more in pollution levies, not less than average. This suggests the desirability of increasing efforts to impose the pollution levy on a larger proportion of the township and village enterprises. Also, recycling a smaller share (e.g. one-half) of charge revenues for pollution control at the paying facilities should further increase the coverage of pollution control efforts. The issuance of discharge permits on the basis of both national concentration standards and a total load allowance (calculated taking in consideration the assimilative capacity of the river) opens the way for trading of pollution allowances.

5 Abstraction Charges and Irrigation Water Pricing

5.1 Policy Context

From the 1950s to the 1970s, under collectivised agriculture, major investments were made in surface water-based irrigation systems to boost agricultural production. These irrigation districts could cover areas of tens of thousands of hectares. However, following agricultural reform and de-collectivisation in the late 1970s, the smaller, village-level organisations of farmers found it harder to raise the capital and coordinate the activities required to take over ownership and then to maintain or extend such systems. As a result, many systems have fallen into disrepair.

In their place, entrepreneurs have established small companies in co-ordination with the village governments that raise capital to sink wells, buy pumps and construct low-pressure underground distribution pipe networks. Farmers then buy water from such an enterprise on a volumetric basis. Private well supplies are often more efficiently managed, as the water suppliers have direct incentives to maintain their assets. Farmers often prefer these sources as being more reliable than district irrigation schemes and offering greater control and autonomy. However, the rural electricity required to operate such systems is subsidised in order to protect farmer income.

This situation of rural water supply entrepreneurs has led to a system under which farmers could be directly paying a volumetric fee for their abstractions. However, with a large number of small abstractions, monitoring, reporting and collection of abstraction charges are patchy. In fact, these abstraction charges (e.g. CNY 0.02 to 0.25/m^3) often end up being levied on the village as a whole and then recharged to the farmers bundled in with other local service charges many months later and often pro-rated by land area, thereby breaking the link between water use and charge. This introduces a free-rider incentive for both the well operator (who is not responsible for the sustainability of the common aquifer but only for his own infrastructure assets) and the farmer, who can benefit by taking more than his share of the commonly administered water supply to boost yields while sharing out the additional costs. Collection of abstraction charges and the allocation of water rights are currently being reformed, pursuant to the 2002 Water Law.

With the abolition of agricultural taxes in 2005, China now has greater flexibility to implement more effective irrigation pricing. In most remaining irrigation districts, fees charged to farmers are much less than the cost of providing the water. Most irrigation supplies are not metered and management systems are vulnerable to abuse of commons, with those who take more than their share benefiting without sanction. Water user associations are being established more widely, pursuant to the 2002 Water Law. These take ownership of the assets and are responsible for setting and collecting user charges for irrigation water. Prices for irrigation water are likely to be much higher than past arrangements.

Due to the inflexible of mandatory means on environmental impact and economic efficiency, economic means becomes supplement, substitute or combined measures for aquatic environmental pollution control. The outstanding advantage of economic incentive programs is, of course, their potential to minimize compliance costs by optimizing cost-effectiveness. Researches and practices on agricultural pollution control have been accomplished in many western countries for a few decades. A series of effective control measurements have been designed and implemented from different views and more incentive policies have been applied. Quantitative frame and evaluation on policy become an approach to get optimized schemes by integrated evaluation of environmental economic benefit. Allocation of pollution loads is made on the basis of gross load control, by economic measures such as tax, subside and pollution trading to eliminate nonpoint source pollution (WANG, 2003).

The most popular environmental policies include pollution taxes, environmental impact assessments, pollution subsidies, and pollution emission permits. GRIFFIN and BROMLEY (1982) found that input taxes, input standards, effluent taxes and effluent standards could achieve socially optimal regulation, but they need more information for some instruments than others. Under conditions of uncertainty, SHORTLE and DUNN (1986) found that management practice incentives, such as incentives for controlling runoff, were superior to standards for management practices, such as standards placed on runoff. The feasibility of point-nonpoint effluent trading in China has been discussed in detail by ZHANG andWANG (2002).

Based on the cause and characteristics of agricultural nonpoint source pollution in China, the scheme of economic policies for control and management of agricultural NPSP is designed in this paper.

- Policy objective. First is to constrain and convert the farmers' unfriendly practices on environment, to encourage and induct sustainable agriculture based on incentive policy. Second is to realize the goal of water quality around watershed with best cost-benefit analysis under the gross-load control of environmental capacity in a watershed.

- Designing principle. Follow the certain environment economic rules and market economic rules with least cost and high efficiency fairly, acceptably, applicably and adaptively.
- System constituent. The key issues include pollution charge, inputs tax for restriction, subsides for induction and incentive, effluent trading for least cost reduction.

The emphases are optimized inputs tax and agricultural chemical tax permit under complete information, as well as sub-optimized inputs tax under incomplete information, subsides for farmer due to positive and negative externality. Different policies have different functions and suitability. In general, tax policy is implemented commonly with prominent effect on restriction, effluent fee and cost apportion are suitable for rural livestock pollution whereas input tax and agricultural chemical tax are favorite for overuse of fertilizer and pesticide. Subside has a good induction and incentive, even it is not so ideal as tax policy, it will promote abatement degree of agricultural nonpoint source pollution by combining with tax and subside. The trading program has provided valuable flexibility for the districts to adjust the initial allocation in response to differences among districts. The limited experience suggests that the trading system has already decreased costs for some districts and provided a formal mechanism for cost-sharing. Transaction costs for these trades presumably have been minimal because the monthly meetings among the districts (which include routine sharing of monitoring information) make it easy to contact and investigate potential buyers and/or discharge targets and measurement system (O'SHEA, 2002).

5.2 Implication

After studying each of these options in detail, some researches concluded that a combination of input fees (tiered pricing of irrigation water) for farmers and tradable discharge permits among irrigation districts required potential advantages over Best Management Practices (BMPs), including the ability to meet a specified pollution discharge goal, regional cost-effectiveness, and administrative ease for both state officials and farmers. Due to vast area in China, there is huge discrepancy on regional natural condition, agricultural production and social and economic development. Restrain approach with tax, such as pollutant and chemical charge to restrict the improper activities of farmers is feasible in advanced watersheds, such as Taihu Lake whereas incentive approaches become dominant in the watershed with least advanced development, such as Dianchi Lake. Tradable market is necessary in which point source pollution has a considerable proportion.

The magnitude of cost savings depends upon factors such as the differences in discharges' marginal costs and transaction costs. While reliable data are limited, a preliminary analysis indicated that the proposed incentive program would provide significant cost savings compared to mandatory BMPs. At least two additional factors make incentive programs attractive. First, based on decentralized decision-making, they preserve the flexibility of individual farmers to respond to changes in economic, environmental, and technological conditions. Second, the programs encourage innovation by providing direct financial rewards for creating better and cheaper pollution control methods.

As discussed above, in different watersheds nonpoint emissions and abatement ability may display different features, and the variance of nonpoint emissions may increase or decrease with the abatement level. In practice, case-by-case study is necessary. In the condition that variance of nonpoint emission decreasing with the abatement level, a trading ratio smaller than one is possible. In the opposite condition, however, for efficient nonpoint abatement, an increase both in the reliability requirement and in variance of nonpoint source emissions would raise the trading ratio, and the optimal abatement allocation would shift toward the point source. Additionally, the optimal trading ratio and abatement allocation also depend on the marginal abatement cost of point and nonpoint sources.

In summary, in the case of diffuse sources of pollution, appropriate regulation must be taken account of in their un-observability and un-verifiability of individual emissions. The applicability of the methods of pollution control depends upon these factors, including the available information, the type of resources to be regulated, the uncertainty, the social cost of damage, the number of polluters to be controlled, and monitoring and transactions costs. Each case must be decided on its own merit.

More remain to be done in the area of nonpoint pollution sources. Apart from management practice regulation and limiting trading programs, the treatment of diffuse sources has remained mostly theoretical. Although these models can be improved upon through the joint work of scientists and economists, there is a demand to implement some of these models in practice to observe how they perform in the field. Many preliminary works are required.

In short, the situation of agricultural NPSP becomes pressing because of the unsuitable and unavailable technology to control the problem. Costs and the administrative difficulty on creating a compliance program were additional stalling factors. This lack of aggressive enforcement was not only due to a lack of legal authority, but because there are too many of them to account for specific pollution-control requirements and administration individually. The same reason has led policymakers nationwide to an apparent impasse in the search to control nonpoint sources of pollution. The crux of this problem is finding a way to: 1) make individual farmers hold responsible for the pollution they generated; 2) maximize cost-effectiveness for farmers and minimize transaction costs for farmers and regulators; 3) make the control program flexible and practical to implement; and 4) simplify the administration.

In conclusion, due to different physical character and social production, different economic approaches are applied in various type areas of nonpoint source pollution. Restrain approach with tax, such as pollutant and chemical charge to restrict the improper activities of farmers is feasible in advanced watersheds, such as Taihu Lake whereas incentive approaches become dominant in the watershed with least advanced development, such as Dianchi Lake. Tradable market is necessary, in which point source pollution has a considerable proportion. The scheme of economic policies for control and management of agricultural NPSP is designed based on the cause and characteristics of agricultural nonpoint source pollution in China. Case study shows that the economic measures are feasible in the watershed based on the consideration on policy, technology, awareness, attitude of local farmers and cost-benefit.

6 Phasing out Farm Input Subsidies

Production and distribution of pesticides and fertilisers are subsidised by the government as an incentive to achieve grain production targets. Price subsidies for fertilisers, chemicals and other farm inputs are estimated at CNY 10 billion (OECD, 2005). They have decreased from more than CNY 30 billion in 1998, when reference prices for fertilisers replaced administered prices, allowing some adjustment for fluctuations in production costs and market demand. However, the 11th FYP proposes to increase subsidies on fertilisers (and road diesel fuel) to promote higher productivity in agriculture.

6.1 Policy Context

Since the beginning of 2004, the Chinese Government has replaced its centuries-old policy of taxing agriculture by a new policy aimed at subsidizing the sector and stimulating rural incomes. The main purpose of this policy is to reduce the growing gap between urban and rural incomes while at the same time promoting grain production. In the following years this policy was further strengthened and expanded, resulting in the concept of 'building a new socialist countryside' as put forward by the Central Committee of the Communist Party in China in 2006.

In the 11th Five-Year Plan, covering the period 2006–2010, building a new countryside takes a prominent place. The goals formulated in this respect in the 11th Five-Year Plan include increasing farmers' income, developing modern agriculture, increasing investment in agriculture and rural areas, and improving the appearance of the countryside (National Development and ReformCommission (NDRC), 2006). Investing in rural areas and stimulating consumptive expenditures by rural households is also an important element of China's 4 trillion yuan stimulus package thatwas successfully implemented in 2008–2009 to deal with the global financial crisis. By raising the purchasing power of 800 million rural people, the Chinese government intends to provide a firm basis for sustained economic growth and to make China's future economic development less dependent on exports.

Another major objective formulated in the 11th Five-Year Plan is the building of a resource-conserving and environment friendly society. The Plan recognizes that two important transformation processes need to be accelerated: First, the change from focusing on economic growth and ignoring the environment towards economic development with an equal emphasis on growth and environmental protection. Second, the transformation from using mainly administrative methods to protect the environment into a comprehensive application of legal, economic, technical and administrative methods to address environmental problems (State Council, 2008).

Governmental policies aimed at controlling natural resource degradation in rural China will be of fundamental importance for the realization of the 'new socialist countryside.' It is generally assumed that the rapid decline in land and water availability and the continued degradation of these natural resource resources is becoming a major bottleneck for further agricultural and rural development. There exists, however, surprisingly little hard evidence on trends in rural land and water availability and quality that is based on consistent definitions and measurement methods and can be used to explore potential future bottlenecks. Natural resource conservation policies in China are traditionally based on direct, centralized regulation and the promotion of state mandated technological improvements (Huang, 2000; Mol & Carter, 2006; Rozelle, Huang, & Zhang, 1997; Xu, Yin, Li, and Liu, 2006). However, recognition of the role of more decentralized policy measures, interventions by more informal, nongovernmental institutions and use of market-oriented instruments play an increasing role in natural resource preservation and restoration in recent years. To support this policy transformation, scientific evidence is needed on the one hand to develop advanced and key frontier technologies that will reduce natural resource pressure and protect the environment. On the other hand, new policies and institutions need to be developed that promote the adoption of such technologies and influence the impact of human behaviour in the desired directions.

The purpose of this paper is to give a systematic overview of major natural resource degradation processes in rural China and to analyse the impact of some recent policy initiatives to address natural resource degradation problems in rural China. Sections 2 and give an overview of recent trends in natural resource use in rural China focussing on land availability and land degradation (Section 2) and water availability and water pollution (Section 3), respectively. In Section 4, two examples of current policy initiatives to address natural resource degradation, namely the Sloping Land Conversion Program and the introduction of new institutions and economic instruments in water management, are discussed and major lessons from these programs are pointed out. The chapter ends with drawing a number of policy implications.

6.1.1 Water availability and quality

The availability of data on water availability and water pollution based on appropriate monitoring methods has also improved in recent years. Official statistics on water availability report a constant renewable water resources volume of 2812.4 billion m3 until the year 1999, but variable and generally lower volumes (presumable taking into account differences in annual precipitation and other factors) since then. Data on water quality in major rivers and lakes are reported since the beginning of the 1990s through an extensive monitoring system set up by the Ministry of Water Resources (MWR) and the Ministry of Environmental Protection (MEP). Data on trends in groundwater levels and quality, however, is relatively scanty. Anecdotal evidence has given rise to much speculation about widespread declines in aquifer levels, particularly in northern China. Evidence on long-term trends in groundwater levels in northern China provided by a joint water sector action program of the World Bank and the MWR, and on more recent trends reported obtained from village surveys held in northern China as well as official statistics of the MWR, make it possible to assess to what extent declining groundwater levels are a reality indeed. In the first part of this section, we will use the currently available statistical evidence to assess changes in the volume of renewable water resources in China and in the volume of water available for use in agriculture. Trends in water pollution in China's major rivers and lakes and in its groundwater resources are the topic of the second part of this section.

Water availability

China is a country with substantial water resources, but due to continued population growth it needs to be shared by an increasing number of persons. The average availability of renewable water resources (surface water and groundwater) in China has declined from 2849 m3 per person per year in 1980 to 1785 m3 in 2009 (National Bureau of Statistics of China (NBS, 2010a), and is rapidly approaching the internationally accepted thresholds for defining water stress (1700 m3 per person per year) and water scarcity (1000 m3 per person per year). Although per capita water availability is slightly higher than in India, it is only onethird of the average of the developing countries and only one-fourth of the world average (Shalizi, 2006).

Due to large differences in precipitation between regions, the distribution of water resources in China is highly unequal. Water availability in the North (757 m3 per person in 2003) is almost 25% below the water scarcity threshold, while water availability in the South (3208 m3 per person) is relatively abundant. Large differences also exist within the northern region, with the so-called 3-H river basins— the Hai and Luan, Huai and Huang (or Yellow) river basins— facing the most severe water scarcity; per capita water availability in the 3-H basins was estimated at 499 m3 in 1999 (World Bank, 2001; World Bank et al., 2001).

Due to the growing scarcity of surface water, groundwater use in agriculture is rapidly increasing. The number of tube-wells used for groundwater irrigation has increased from 0.2 million in 1963 to 4.7 million in 2003 (Zhang, Wang, Huang, & Rozelle, 2008) and to 5.2 million in 2007 (Ministry of Water Resources (MWR), 2009). Nearly all these tube-wells (95%) are in northern

China, even though only 30% of the groundwater resources are located in the northern part of the country (Wang, Huang, Rozelle, Huang, & Blanke, 2007). Using the results from a regionally representative village survey, Wang et al. (2007) examined the impact of groundwater extraction on the water table level in northern China. They found that the water table had increased between 1995 and 2004 in 16% of the 448 villages that were surveyed. In 35% of the villages the groundwater level showed little or no decline since the mid-1990s, while in 48% of the villages the water table had declined. In 8% of the villages, the rate of decline exceeded 1.5 m per year, implying 'serious overdraft' (following the definition of the MWR). Official statistics for the North China Plain (MWR, various years) show that during the period 2000–2007 the groundwater level on average declined in 61% of the monitoring sites while the level increased in the remaining 39%. The total groundwater volume increased in two out of 8 ears (2003 and 2005; years in which precipitation was relatively high), while it declined in the other 6 years.

Evidence presented in World Bank et al. (2001) for the 3-H basins in northern China shows that groundwater depletion is most severe in the Hai basin. Between 1958 and 1998, shallow groundwater levels have declined between 10 and 50 m in a vast area surrounding Beijing, Shijiazhuang and Tangshan. In all four subareas of the Hai Basin, the use of groundwater exceeded the amount of exploitable fresh groundwater in 1997; in the Huai and Huang basins this was the case for two of the 15 subareas (World Bank et al., 2001: fig. 3.11 and table 3.9). Groundwater depletion also takes place in areas where authorities do not supply safe surface water due to growing water pollution, such as the lower reaches of the Yangtze. It is estimated that 25 billion m3 of nonrechargeable deep-aquifer groundwater were mined in China in 2000, mainly for agricultural purposes (World Bank and State Environmental Protection Agency (SEPA), 2007).

Total water use declined from 556.6 billion m^3 in 1997 to 532.0 billion m^3 in 2003, but increased since then to 591.0 billion m3 in 2008. The water available for use in agriculture has been reduced by the higher water demand for industrial and consumption usage, which increased by 29.2% over the period 1997–2008. By using more efficient irrigation systems and cultivation methods, total water use in agriculture declined by 12.4% from 392.0 to 343.3 billion m3 between 1997 and 2003 despite an increase in the irrigation area by 5.4% during this period (World Bank, 2006). After 2003, however, water use in agriculture increased by 6.7% to 366.3 billion m^3 in 2008, while the irrigation area increased by 8.3% during the same period (National Bureau of Statistics of China (NBS), 2003–2009). As a share of total water use, the use of water in agriculture has steadily declined from around 80% in 1980 to 62.0% in 2008 (National Bureau of Statistics of China (NBS), 2010b; World Bank, 2006).

Water pollution

Not only availability, also pollution of available fresh water sources is a major cause for concern. Water quality monitoring data for the seven main rivers in China during the period 1991–2008 show that water quality is significantly lower in the rivers in northern China as compared to those in the South (MEP, various years; World Bank, 2001, 2006). The Liao and Huai rivers, and especially the Hai River, in northern China suffer in particular from heavy pollution. The much smaller water flows (and hence their smaller assimilative capacity) in the rivers in northern China is an important factor explaining this difference. Other factors include the relatively low population pressure in some provinces in southern China and the fact that industries tend to be concentrated much more in the lower reaches of the river basins in the South (World Bank, 2001). The quality monitoring data further show that water quality has improved between 1990 and 2008 in the South (that is, in the Yangtze and the Pearl River) although these rivers still contain areas of very poor water quality, particularly in their tributaries (World Bank, 2006). Water quality further deteriorated in the rivers in the North during the period 1991–2005, particularly in the Hai and Huai river basins (World Bank, 2006; Xie, 2009). Since 2005, however, water quality has also steadily improved in the major river basins in the North (MEP, various years). Currently, water in around 40% of the monitoring sites in the North is suitable for human consumption after treatment as compared to 85% in the South (Ministry of Environmental Protection (MEP), 2009). The rural population relies primarily on surface water as the main source of drinking water, and is therefore more vulnerable to possible pollution than people living in the cities who have access to alternative sources of drinking water. The share of industrial wastewater in total water pollution has declined since the mid-1990s due to successful treatment of industrial wastewater.

Untreated domestic wastewater discharge has become the most important pollution source since 1999, while non-point source pollution, primarily caused by fertilizer and pesticide runoff from farmland and infiltration of livestock waste, is becoming increasingly important (Xie, 2009).

Among the 28 key lakes and reservoirs under the national monitoring program, only 6 reported water quality suitable for human consumption after treatment in 2008 (Ministry of Environmental Protection (MEP), 2009). Eutrophication is a major problem in many of the lakes and reservoirs in China, pollution by nitrogen (N) and phosphorous (P) being the main cause. Efforts to control eutrophication in the three most critically affected lakes, the Dianchi Lake (Yunnan), Chao Lake (Anhui) and the Tai Lake (Jiangsu–Zhejiang), have met with limited success until now (see Table 5). Water quality monitoring data for these three lakes show that the quality of the Tai Lake improved between 1992 and 1998 but greatly deteriorated between 1998 and 2006, resulting in a major algae bloom overtaking the lake in May 2007. Water quality in the Dianchi Lake steadily deteriorated throughout the entire 1992 - 2006 period, while it improved in the Chao Lake except for an increase in Total P between 1992 and 1998. Around 70% of the nutrient inflow into the Chao Lake and the Dianchi Lake is derived from agricultural runoff (Shalizi, 2006: p.11).

In around 50% of all regions, shallow groundwater is polluted by wastewater discharges from industrial, municipal and agricultural sources (Xie, 2009). There is some anecdotal evidence of declining trends in groundwater quality, but lack of comprehensive time-series data makes it impossible to draw general conclusions. According to World Bank (2001), however, there seems to be little doubt that groundwater quality is deteriorating, particularly in the neighborhood ofmajor cities and in aquifers that are close to the surface. In coastal areas, falling groundwater levels due to over-pumping (see above) causes migration of poor-quality groundwater into good-quality aquifers and causes intrusion of salty seawater in coastal regions. Salt water intrusion in coastal aquifers is found to be common in some 72 coastal areas covering a total area of 142 km2 (World Bank et al., 2001).

6.1.2 Rural water issues

The information presented in this section shows that use of water for industrial purposes and domestic consumption is increasingly reducing the amount of water available for agriculture in northern China, particularly in the 3-H river basins. In response, farmers resort to water-saving irrigation systems and cultivation methods and to the use of groundwater. Groundwater tables have fallen considerably in the Hai river basin, but evidence on other parts of northern China is mixed. Water pollution is a major problem in the rivers in northern China, particularly in the Hai River, and was getting worse until around 2005. Water quality problems in the major rivers in southern China, on the other hand, are less severe and getting less. Pollution is also a major problem and is getting worse in the Dianchi Lake and the Tai Lake in southern C northern China as compared to those in the South (MEP, various years; World Bank, 2001, 2006). The Liao and Huai rivers, and especially the Hai River, in northern China suffer in particular from heavy pollution. The much smaller water flows (and hence their smaller assimilative capacity) in the rivers in northern China is an important factor explaining this difference. Other factors include the relatively low population pressure in some provinces in southern China and the fact that industries tend to be concentrated much more in the lower reaches of the river basins in the South (World Bank, 2001). The quality monitoring data further show that water quality has improved between 1990 and 2008 in the South (that is, in the Yangtze and the Pearl River) although these rivers still contain areas of very poor water quality, particularly in their tributaries (World Bank, 2006). Water quality further deteriorated in the rivers in the North during the period 1991–2005, particularly in the Hai and Huai river basins (World Bank, 2006; Xie, 2009). Since 2005, however, water quality has also steadily improved in the major river basins in the North (MEP, various years). Currently, water in around 40% of the monitoring sites in the North is suitable for human consumption after treatment as compared to 85% in the South (Ministry of Environmental Protection (MEP, 2009).

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6.2 Initiatives to reverse water degradation in rural area

The growing awareness of environmental and ecological issues in China is reflected in the higher priority the government attaches to these problems, and, equally important, the increased willingness to tackle them. After many years of high economic growth, the ability of the government to act has also gradually improved thanks to its growing financial capacity, especially at the central level. These developments have facilitated new government activities to combat soil erosion and other resource degradation problems on a large scale. In this section, the Sloping Land Conversion Program (SLCP) and water scarcity management policies are analyzed as two typical cases of current policy initiatives to reverse resource degradation trends and improve resource efficiency.

The SLCP is one of the world's largest programs offering ('supply-side') payments for environmental services (PES) in terms of scale, payment and duration (Liu et al., 2008). It combines the traditional top-down approach in environmental and natural resource management with economic incentives aimed at changing farm household enterprise choices. Recent policy initiatives and pilot projects on water scarcity management introduce institutional innovation as well as resource pricing and ('market-oriented') PES. We discuss each of these two cases in turn.

6.3 Management with EPIs

Market-oriented instruments as well as institutional innovations have been introduced in recent years in the management of China's limited water resources. They include water pricing, PES, and the introduction of water user associations.

Water pricing has been introduced in agriculture (and other sectors) to increase water use efficiency. Before the economic reforms started in 1978, water was generally considered a free good. By the end of the 1970s, irrigation delivery efficiency (the ratio of the water actually taken up by the crop to the amount of water diverted from the source to the area) was only 0.3 (Liao et al., 2008). Water fees were gradually introduced and increased since then in an effort to meet the cost of water supply and improve water efficiency. In 1997, the Water Irrigation Industry Policy (Shui Li Chan Ye Zheng Ce) was issued by the State Council stating that the water price for irrigation schemes should fully recover all water supply costs, including debt service, taxes, and a reasonable profit margin (Jia, & Jiang, 1999; Liao et al., 2008). Current levels of water prices, however, are still insufficient to recover water supply costs. Water fees accounts for less than 40% of total water supply costs in 100 large and medium-size irrigation districts in 2002, and collection ratios of water fees average only 50–60% nationwide (Liao et al., 2008; Ministry of Water Resources (MWR), 2002a, 2002b).

On 1 January 2004, a new water pricing regulation covering all economic sectors was introduced. Its main objectives are to increase the water price so as to fully recover water supply cost and to treat water as an economic good rather than being administrated as an institutional fee. However, there is still controversy over the price to be charged for irrigation water as higher prices may seriously affect two other major national policy goals, namely reducing the rural–urban income gap and promoting near-self-sufficiency in food production. Moreover, the absence of infrastructure to monitor surface water use in many regions and the option to substitute self-provided groundwater for system-provided surface water pose important limitations of the use of water pricing in agriculture (Liao et al., 2008; Liao, Giordano, & de Fraiture, 2007). Recent interviews by the authors of this paper with water supply authorities in Minle County, Gansu Province reveal that water prices are still decided upon by local governments, have not been increased over the past 10 years, and cover around one half of the supply costs. Huang, Rozelle, Howitt, Wang, and Huang (2010) find that the current cost of groundwater is far below the true value of water in northern China. Their results indicate that doubling the water price (by adding a tax or fee to the price of electricity) would cause a reduction of 20% of the current level of groundwater use in rural Hebei Province, but this water saving would be achieved at the expense of grain production and rural household incomes.

An interesting recent phenomenon is the emergence of groundwater markets in northern China. According to survey data,15 in 1995 only 9% of interviewed villages knew groundwater markets, but by 2004 there were groundwater markets in 44% of the sampled villages (Zhang et al., 2008). Traded groundwater is supplied by tube-wells owned by private farm households or groups of individuals. In these groups each member has usually a share proportional to the investment stake in the tube-well. The emergence of groundwater markets also meant changes in the property structure of tube-wells and other institutions. Before 1980, most tube-wells were owned and operated by collectives. In the early 1980s, the property structure of many tube-wells started to shift to private ownership (Shah, Giordano, &Wang, 2004; Wang, Huang, & Rozelle, 2005). By 2004, as much as 70% of the tube-wells were privately owned (Zhang et al., 2008).

Concurrent with the changes in water pricing policy, two major reforms took shape: (1) The introduction of water property rights and of systems of PES, and (2) the setting up of water user associations (WUAs) and independent water supply units. Water property rights have been assigned in some pilot areas such as the Yellow River Basin, the Hei River Basin in Gansu Province, Yiwu City in Zhejiang Province, and in Beijing Municipality and Hebei Province. The experience with water allocation and quota systems in the Yellow, Hei and other river basins is reflected in a new document for water allocation rules issued by the MWR (Ministry of Water Resources (MWR), 2007). In the Yellow River Basin, water quotas are assigned to different regions by the Yellow River Management Committee. The resulting water quotas can be traded between different regions (Hu & Ge, 2004). Water quotas are allocated by the local government to water users in Zhangye City since 2002 as one major element of a pilot project on 'Building aWater Saving Society' initiated by the MWR.16 Its purpose is to save water from agricultural use in Zhangye City for low-reaches ecological use in Inner Mongolia and to increase agricultural water use efficiency (Office of Building Water Saving Society in Zhangye City (OBWSS), 2004). Despite claims made by policy makers and local leaders (e.g. Liu, 2006; Zhao, 2007), however, trading in water use rights has virtually been absent (Zhang, 2007; Zhang, Zhang, Zhang, & Wang, 2009; own observations). Transaction costs as well as management, legal, administrative and fiscal barriers hinder the development of water markets in the region. Farm households prefer to use groundwater to deal with water shortages over buying user rights from others. And local governments, who are supposed to buy surplus water back from farmers at higher prices, are short of financial resources to do so and encourage farmers to use all disposable water.

The exchange of water use rights between Dongyang City and Yiwu City, located along the Jinghua River in Zhejiang Province, is an interesting example of successful water rights trading between regions (the first of its kind in China). Water from a reservoir in Dongyang City, used for agricultural irrigation, is being sold to Yiwu City as drinking water (Zhao & Hu, 2007). This example shows a successful case of payment for water services (PWS). Whereas programs such as the SLCP rely on available government funds and direct them to ecological recovery activities, PWS and (more generally) PES schemes create a new market to increase funding and target those funds to water saving and other conservation activities. In such schemes the services providers and users decide on the quantity to be traded and its price, instead of the (central) government. A higher level of social welfare will be the outcome, provided appropriate institutional arrangements can be designed for facilitating the transfer of funds (Dixon & Xie, 2007).

A similar initiative is being developed between the governments of Beijing and Hebei Province with the purpose to stimulate water-saving measures in the area upstream of Miyun reservoir, a major drinking water reservoir for the citizens of Beijing (Guo, 2007; Zheng & Zhang, 2006). Research by Zhao and Hu (2007) on the Dongyang-Yiwu PWS system shows that trading water use rights from agricultural to non-agricultural use did improve water use efficiency. A lack of supplementary policy measures that would promote the participation in the decision making by upstream farmers, however, failed to encourage most upstream farmers to switch to water-saving crops. These farmers experience substantial income losses from the higher water prices that result from the water trading reform, and are not being compensated. Both the property right of the reservoir and the amount of water it contains are not clearly defined, which complicates the payment of compensations to upstream farmers.

A major institutional innovation in irrigation water management has been the introduction of WUAs. Starting in 1992, they were introduced by the World Bank in Hunan and Hubei Provinces, and later in Xinjiang Province to manage water resources at the local level (Yu, 2007). The MWR subsequently disseminated the WUA approach as a good practice throughout the country (Lin, 2003). By 2006, around 10% of the villages in northern China had adopted WUAs (Wang et al., 2010). WUAs act as the buyers of water from water supply institutions, coordinate delivery at the local level, organize water guards for water monitoring, collect water charges among its participants, and organize canal and facility maintenance. To be successful, WUAs should satisfy five key principles: There should be adequate and reliable water supply, the WUA should be organized hydraulically (not administratively), election of leaders, management of the WUA and decision making should be with the farmers (without local government interference), water should be charged volumetrically (not according to land area), and the WUA should have the right to collect water fees (Lin, 2003; Wang et al., 2010; Xie, 2007).

An empirical study among WUAs in Ningxia, Gansu, Hubei and Hunan Provinces by Wang, Xu, Huang, and Rozelle (2006) and Wang et al. (2010) finds that there are important differences in the extent to which the five key principles are implemented, and that the degree of implementation has important implications for water use efficiency. Water use in rice, wheat and maize in World Bank-supported WUAs, which mostly operate according to the five principles, is found to be 15 - 20% lower than in traditionally managed villages. In villages where participation by farmers plays only a minor role and water management reforms have been only nominally implemented, the establishment of WUAs has had little effect on water use. The study further finds that crop yields and incomes are not significantly different between World Bank-supported WUAs and other WUAs.

6.4 Effectiveness surveys

We can conclude from this overview of studies evaluating the SLCP and recent water scarcity management policies that the ecological effects of both are generally positive. The limited attention paid to differences in local conditions in the SLCP (a ‘supply side’ PES) and the apparent lack of volunteerism in many regions, however, limit the potential ecological and efficiency gains of such a program. Moreover, prevailing bottlenecks in land and labor markets limit structural shifts of labor towards off-farm activities and thereby undermine the long-term sustainability of the SLCP.

Pilot projects on introducing ‘market-based’ PES between local governments have been successful in introducing water savings in upstream areas. Problems remain, however, in passing the benefits on to the actual suppliers of the water services, the upstream farmers. Water pricing is introduced to stimulate water savings in agriculture.

But surface water prices are still set by the government at levels below the actual cost price in order to meet other rural development goals (poverty reduction, food self-sufficiency) and environmental goals (maintaining groundwater levels). A major institutional innovation is the introduction of WUAs to stimulate water management at the local level. Limited farmers'participation in decisions making, however, seems to limit the potential benefits of such local resource management organizations in several regions at the moment.

6.5 Implication

In response to the global food crisis in 2007-2008, the Chinese Government reiterated in November 2008 its goal of stabilizing the country's grain self-sufficiency at a rate above 95% and attaining a capability of producing 540 million tons of grains by the year 2020 (National Development and Reform Commission (NDRC), 2008). During the period 2007–2009, the average annual grain production stood at 520 million tons (with a self-sufficiency ratio of 97%). Hence, a production increase of around 4% will be needed during the coming decade to meet this goal.

The natural resource base needed to support this grain self-sufficiency policy is relatively small and under continuous pressure. Cultivated land per capita is only one-third of the world's average, while the amount of water available per head is one-fourth of the global average. Almost 40% of the land suffers from (water and wind) erosion, 90% of the usable grassland in China is degraded to some degree, and water in 80% of the major lakes and reservoirs and 45% of China's main rivers is unsuitable for human consumption after treatment.

Despite the weak natural resource base, production of food grains increased sufficiently in recent years to keep up with China's growing population. Increases in yields per hectare of land and per cubic meter of water have been sufficiently high to offset the declines in land and water availability and the various degradation processes affecting land and water quality. Whether or not these successes can be sustained in the near future will depend to a large extent on the ability of the government to maintain its resource base and to reverse some of the worst resource degradation processes before they start to have significant negative effects on crop yields.

China's cultivated land area has steadily declined since the onset of the economic reforms, to a level of 121.7 million hectares in 2007. We find that ecological recovery programs have greatly accelerated the decline in cultivated area in the beginning of the new century. Conversion of farmland into land for urban use also played an important, but much smaller, role. To ensure grain self sufficiency and economic and social stability, the State Council announced in 2008 in its Land Use Plan 2006–2020 that the country's cultivated land should remain at 121 million hectares by 2010 and at 120 million hectares by 2020 (State Council, 2008).

To realize these goals, expansion of the SLCP was slowed down (Ministry of Land and Resources (MLR), 2007, 2010) and ended in 2009 before the program had fully reached its intended scale. Yet, ongoing urbanization may make it difficult to realize the land preservation goals. Each year, around 0.2 million hectares of cultivated land is converted into (mainly urban) construction land (Table 1; Ministry of Land and Resources (MLR), 2007, 2010). With continued rapid economic growth and the recent policy emphasis on developing small and medium-sized cities in rural areas (as emphasized in the so-called Number One Central Document of 2010), these rural–urban land conversion trends are likely to continue if not to intensify. The results from an analysis of China's urban expansion during the period 1995–2000 by Deng, Huang, Rozelle, and Uchida (2009) indicate that 10% GDP growth causes around 3% urban expansion.

The recent 'land for land' policy, which requires local governments to reclaim the same amount of arable land before existing arable land is allocated for non-farming purposes, may be an appropriate instrument to reach the goals as specified in China's Land Use Plan 2006–2020, if it can be implemented effectively. Average land productivity, however, is expected to decline when highly productive land in the urban fringe is replaced by land reclaimed elsewhere (Deng et al., 2006; Tan et al., 2007). Introducing more market mechanisms into rural–urban land conversion may be another appropriate policy option (Tan et al., 2011-This volume).

This option is not only expected to reduce the current over-conversion of farmland, but will also have positive welfare effects on rural households that are affected by this conversion. Land degradation remains a major cause for concern. Large-scale ecological recovery programs have been successful in increasing the size of forest land and in reducing water erosion. But there is also convincing evidence that the 3NSCP and the SLCP contributed to a worsening of wind erosion in (semi-)arid regions (Cao, 2008), although the magnitude of this impact is unclear.

Instead of having a strong focus on tree planting, these programs should aim at restoring natural ecosystems (such as natural steppe and grassland vegetation) in these regions (Bennett, 2008; Cao, 2008; China Council for International Cooperation on Environment and Development (CCICED, 2002). Moreover, the apparent lack of volunteerism in many regions and prevailing imperfections in land and labour markets limit the potential efficiency gains and long-term viability of the SLCP. Measures announced during the CPC Plenum in 2008 to assign more land property rights to rural households and to gradually integrate socio-economic policies for rural and urban households, are an encouraging step towards the elimination of remaining factor market imperfections, and may thereby contribute to more sustainable land use.

The amount of water available for agricultural production has steadily decreased as a result of increased competition with other sectors, and is expected to further decline in the coming decades. In response to the growing scarcity of surface water, farmers in northern China, where water scarcity is most eminent, resort to water-saving irrigation systems and cultivation methods and to the use of groundwater. Groundwater tables have fallen considerably in the Hai river basin as a result, but evidence on other parts of northern China is mixed. Significant additional water savings can be achieved through expansion of WUAs (which cover about 10% of the villages in northern China at the moment), provided that member households actively participate in decision making and that other basic principles of collective action are satisfied.

More attention may also be paid to collective action options in addressing another major problem, rangeland degradation. Grassland improvement programs in the 1980s focused on assigning long-term user rights to individual herder households, but traditional community-based management systems can still be found in many regions (Banks, Richard, Ping, & Zhaoli, 2003; Nelson, 2006). More recently, a program 'converting pastures to grasslands' is being implemented which bans grazing (either permanently, temporary or seasonally) in specific zones and stimulates herders to take up sedentary, town-based lives (Yeh, 2005). Given the spatial and temporal variations in climate in (semi-)arid regions and the need to move herds in response to environmental risk, community-based management may have socio-economic as well as ecological benefits over household based management (Banks, 2003; Nelson, 2006; Ngaido & McCarthy, 2005).

Pollution of river water has diminished in southern China during the last two decades and in northern China since about 2005, due in particular to the successful control of industrial wastewater. Non-point source pollution, primarily caused by fertilizer and pesticide runoff from farmland and infiltration of livestock waste, however, is becoming an increasingly important source of water pollution. It is the main cause of the pollution in Dianchi Lake, Tai Lake and several other lakes and reservoirs, and also a significant contributor to the pollution of river waters. Stimulating off-farm employment, particularly migration, may help to reduce such non-point source water pollution (Shi et al., 2011).

Increasing the price of water and fertilizers towards their true (social) value may be an effective way to address water scarcity and (agriculture-based) water pollution. After more than a decade of marginal price increases, fertilizer prices increased by 38% in 2008 as a result of soaring global prices (NBS, various years). Moreover, government control over fertilizer prices (except potash) has been removed since the beginning of 2009, leaving the prices of fertilizers to be decided by the market. A thorough evaluation of these recent developments can provide important insights into the extent to which fertilizer (and water) price reforms are able to achieve environmental goals without jeopardizing two other major policy goals, namely remaining self-sufficient in grains and reducing the rural–urban income gap.

7. Pricing Irrigation Water

In 2000 at The Hague, the World Water Council adopted as a 'Vision' the proposition that water should be charged at full cost to all users. There is, however, a variety of ways of pricing water.

Water in a river is a resource, delivered to farmers via an irrigation infrastructure. The price that Chinese farmers are charged for the water they use on their farms typically combines a resource fee and an infrastructure charge. A resource fee seeks to capture the opportunity cost of water in a river in its best alternative use (which may include environmental flows). An infrastructure charge is the fee charged for delivering water from the river to farmers'fields, including the capital cost of constructing, operating and maintaining an irrigation system.

In China as in most places, such prices are set by the state, though in principle private organizations could be given this right. The charge to farmers for irrigated water can be set in several ways:

- Area: either a fixed price per hectare of irrigated land (perhaps with a quota) or different fixed prices per hectare of 'subsistence' land and 'above-subsistence' land;
- Crop: avariable price, depending on either the crop grown or the season (or both), possibly with a lower price for subsistence crops;
- Volumetric: a fixed or variable price per unit of water delivered to a farm;
- Multipart: volumetric pricing at the level of, say, a village, combined with area

or crop pricing within the village (Hussain, 2005, pp.63 - 65).

All these prices tend to restrict the aggregate amount of water that farmers use. In principle, though, the most effective way of restricting the use of water by irrigation farmers is to charge volumetric prices for the water they use. Since irrigation infrastructure has high fixed costs, volumetric charges may need to be supplemented by fixed fees (see Johansson, 2000).

A special form of volumetric price is a price equal to the marginal cost of supplying water, including the social (scarcity) value of water in a river, infrastructure and maintenance, and administration. If farmers are rational and have perfect information, then in theory they produce the highest return for the amount of water supplied (Johansson, 2000). If this price is the same across all sectors, inter-sectoral allocations are efficient in this sense, too.

Markets can determine prices, under particular conditions (Dinar andMody,2004; Easter etal.,1997). First, users of water from a river have rights to a certain volume of water per period – a year, a quarter or a season – separate from the right to use land.

Second, the right to some or all of the water can legally and in practice be sold to others (by 'inpractice' we mean that the water can actually be moved from the seller to the buyer).Third, a management system exists that resolves conflicts over external effects (such as return flows and pollution), fluctuations in river flow and the rights of in-stream users. Finally, rights and contracts are legally enforceable and trading is not too costly. If there are no externalities from the use of water, all parties are fully informed, there is complete certainty, competition is perfect and there are non-increasing returns to scale, then water markets efficiently allocate water: the price paid by users equals the marginal cost of supply and equals the marginal productivity of water for users (Dinar and Mody, 2004; Johansson, 2000). If such a water market exists, water prices are determined by it.

Even if some of these conditions are not met, a rights-based system of water transfers could, it is argued, equitably improve the efficiency of water use (Easter etal.,1999) - that is, values of output per cubic metre could be raised. Users have an incentive to conserve water, for the saved water can be sold, increasing the supply of water to locations and sectors where demand and ability to pay are greatest. The system is equitable because those whose consumption decreases (and so whose production falls) are compensated by the sale of water. If the sum of all use rights (including environmental flows) is equal to or less than supply, and if rights are adjusted to natural variations in supply, then a tradable water rights system is also sustainable (Easteretal., 1998,1999; Rosegrant and Binswanger,2004). In practice, the costs of designing and administering a transfer market increase as the number of rights-holders increases, so the scale of allocations is important: use rights and transfer rights could be assigned to provinces, counties, townships, villages, or individuals, which could then sub-allocate rights to users within their domain - multiple markets at different scales under various jurisdictions are possible.

Thus, the first claim is that if water prices are quasi-volumetric and near to marginal costs, then water use is efficient. A central body can set these prices, but a system of tradable water rights also achieves such benefits. Secondly, if prices are approximately equal to costs, they facilitate appropriate levels of investment and maintenance. Related to this claim is a third: enhanced investment and maintenance improve water services to farmers – thus the broad package of price changes does not harm the poor. As the Asian Development Bank puts it: the evidence from scores of water projects is that the poor are increasingly willing to pay for services that are predictable and effective (ADB, 2003,p.25; see also Hansen andBhatia,2004). Thus, it is argued, reforms do not harm poor consumers and often improve their access to water (ClarkeandWallsten,2002). Higher charges for irrigation water help recover the cost of providing water delivery service; give an incentive for efficient use of scarce water resources; and act as a benefit tax on those receiving water services, providing resources for further investment to the benefit of others (Perry,2001; see also Cai and Rosegrant, 2004), all without affecting equity (TsurandDinar,1995).

In China, actual prices charged for irrigation water are thought to be well below levels that are efficient (i.e., that markets would set). Prices do not even cover the full costs of operating and maintaining irrigation systems (Hussain, 2005; Wangetal.,2004; Yangetal.,2003). The price of irrigation water varies across the country, being generally lower in the south (RMB0.02–0.03/m^3 in provinces like Guangdong and Chongqing) than in the north (RMB0.10–0.14/m^3 in provinces like Gansu and Shanxi) but is generally one half-one third of the cost of supply (Zhou and Wei, 2002). Other estimates differ, but also claim that existing prices are less than costs (Shi and Xu, 2001; World Bank, 2000b), despite steep increases in the price of water in the Huang He basin in the last five years (YRCC, 2001). A computable general equilibrium model estimates that the market price for water in China should have been nearly RMB4.00/m^3 in 2000, about thirty times the highest prices now paid for surface water by irrigation farmers (He andChen,2004). Partly as a consequence of these differences between the price and the cost (or the value) of water, various actors in China's complex water resource management structure, including the Ministry of Water Resources, are considering the merits of a system of transferable water rights (Lohmar et al., 2003). Indeed, in principle the establishment of water markets is now possible after the revised national Water Law came into force in late2002 (Yuan and Chen,2005). In 2003 the YRCC established five pilot water right transfer projects in Ningxia and Inner Mongolia (Yuan and Chen, 2005), permitting factories to invest in water-saving irrigation technology and buy from irrigation districts the water thus saved.

Yet these arguments, that tradable water rights or marginal cost pricing are the key to successful management of water, are applied in country after country in Asia and elsewhere; they are simply replicated in China by the ADB, the World Bank, foreign experts, western-trained Chinese economists and engineers, and the central agencies of water management in China (such as the Ministry of Water Resources). The identification of problems is the same for China as for the UK or the USA; the solutions proposed are the same too, despite their different physical, economic, social, legal and institutional conditions.

The water problems in China may indeed be similar to those in western Europe; however, solutions that work efficiently in western Europe may be inefficient or ineffective in China (Biswas,2001). We now turn to consider these different conditions.

7.1 Framing the culprits

Marginal cost pricing and water trading work to increase technical and allocative efficiencies only if the practices of farmers are the principal reason for water being used in efficiently in China. The language is one of inefficient farmers – they use flood irrigation rather than drips and sprinklers; they grow low value crops with water; they apply too much water; they do not time water applications properly. The story is repeated in the Chinese press (Yu,2004), among irrigation engineers and plant breeders (Kijneetal.,2003; Tuongand Bouman,2003) and among economists (Cai and Rosegrant,2004). Some commentators are more sympathetic, arguing that water-saving irrigation practices and technology are not widely used because there are

no incentives in place for farmers to benefit directly from saving water (Lohmar etal.,2003). The result of this discourse is development projects that seek to change irrigation practices on farms - such as the World Bank's water conservation project on the north China plain (Beijing, Hebei, Liaoning, Shandong) (for details see World Bank, 2000a, b). The implication is clear: many, if not most of the problems lie at the farm level and are the fault of farmers.

However, the management of surface water for irrigation reveals a different story. We observe a system that is in chaos. The problem, perhaps is not so much resource scarcity as scarcity of appropriate management (Yang and Zehnder, 2001). At an institutional level there exist the confusions and uncertainties associated with the tiao-tiao-kuai-kuai structure of governance inChina. The detailed management of irrigation districts also provides little scope for farmers to modify their irrigation practices.

The institutions that manage surface water in China are many and complexly linked (Hou, 2000; Lohmar etal.,2003; Zhou andWei,2002). Consider the Shandong Bureau of the YRCC, with which we have worked. It 'belongs' to the YRCC, one of a set of national river basin commissions, whose primary task is to approve and enforce provincial water withdrawal plans. Inturn, the YRCC is an agency of the Ministry of Water Resources, which is itself responsible for planning, constructing, and managing all water-related projects including flood control, power generation, water transport, domestic water treatment, and industrial water use in addition to irrigation; the bureaus under the Ministry are responsible for managing plans of higher level agencies and for administering water resource systems. In other words, a vertical (or tiao tiao) structure channels commands from State Council through central government ministries (such as the Ministry of Water Resources), to water resource bureaus at provincial, prefectural, county and township levels. Individual irrigation districts report to their appropriate water resources bureau.

At each level of government, the management of water also requires cooperation between the different agencies that are responsible for different functions: this is horizontal cooperation (or kuai kuai). The line of power flowing from the Ministry of Water Resources is supplemented by lines from the State Environmental Protection Administration (wastewater, pollution), the Ministry of Geology and Mining (groundwater), the State Price Bureau (prices), the Ministry of Construction (urban water delivery) and the Ministry of Agriculture (on farm technologies).

These agencies and their subordinate bureaus must cooperate within the central government and within each province, prefecture and county. At each level, goals for social and economic development are set by the government at that level, into which must fit the activities of the functional agencies.

The Shandong Bureau is thus located in a matrix. Power flows from high ranking agencies (YRCC) and a demand for cooperation flows from equivalent ranking agencies in the same government (Shandong Province). This system generates conflicts between different users (urban and rural consumers are served by different agencies); between different sources of supply (ground and surface water are administered separately); between supply and conservation; between national, basin, provincial and local levels of government; and between flood protection (prevention), water supply and pollution control. Some conflicts are delicately described by Liao and Xiao(2005). In many respects, this system operates to force irrigation farmers to use as much water as possible.

Consider the level of provinces. The Shandong Bureau, for example, tells us that there is a stated allocation of water to provinces; but the allocations are periodically revised and subject to exceptional circumstances (such as drought). Since the Bureau does not receive as large an allocation as it seeks, it competes with other Bureaus for water and, fearing that it will use some of its allocation in the future, uses all of (or more) of its allocation to ensure that the same amount is allocated next year. Other Bureaus are in the same competitive game. Down within Shandong, we see a similar competitive ethic forcing irrigation districts to use all their allocations too.

Consider also the level of an irrigation district. In some irrigation districts, managers are given explicit incentives to conserve water (Wangetal.,2003). In many, though, the procedure is like that we observe within Jinan municipality, Shandong, Local governments can tell the Shandong Bureau how much water they want and when; yet the amount and timing of releases from the Huang He into the irrigation canals are entirely at the discretion of the YRCC. Water flows into an irrigation canal for a fixed period that is determined by the YRCC and flows through the distribution canals to the last farmers in the district.

Farmers have to use water when it is made available, even if they do not need it. The first farmers along the distribution canals use what they want or need; some is then lost to seepage; the rest is available to the next farmers along the canal.

As we see these farmers irrigate, they do not have the option of being more efficient users of water. There is no water storage in the system, so water applications cannot be timed; farmers cannot reduce applications now to use more later. Since farmers are charged for water on an areal basis, any reductions in their use of water translate into yield reductions but no monetary savings.

In other words the language of blame needs to be re framed: the inefficiency lies in the system of water management. The YRCC gives no managers or farmers incentives - indeed, gives them little opportunity - to save water.

7.2 Taxing Farmers

There are several reasons why the price elasticity of demand for irrigation water is so low. First, at very high levels of application of water, more water reduces yields (Fraiture et al., 2002). Beyond a certain point water is subject to negative marginal returns, not diminishing marginal returns. Even at very low prices, farmers may not use all the water they can get. Thus, if prices are low, an increase in price may have little effect on the use of water, since it is not prices that constrain use but the shape of the production function. Secondly, farmers may be subject to legal or de facto water rationing, which restricts their applications of water to sub-optimal levels (Fraitureetal.,2002). Such rationing includes canal dimension and pump capacity, and means that the use of water in Chinese agriculture is in aggregate well below the demand curve (Ehrensperger,2004). Again, this means that water is rationed by command rather than by price, and moderate changes in price may not influence farmers'decisions. Thirdly, though, the price charged for water is only one component of the cost paid by farmers for water. Standard components include the water resource fee and the fees paid for delivery (infrastructure capital costs, operations and maintenance).

In addition, when water is drawn from a river on the north China plain, the water in irrigation channels is below ground level and is gravity fed, so farmers have to pay to pump the water up onto their land. For example, we observed farmers in Beidong village (Jinan, Shandong) paying an average of about RMB5.00 per mu per year for water; but they paid another RMB95.00 per mu per year for diesel to pump that water from the canal and another RMB60.00 per mu for the pumps (amortised over five years). In other words, estimated price elasticities are not for resource and infrastructure fees but for the very much larger resource plus infrastructure fees and pumping costs. Such data prompt three conclusions about the costs of irrigation water in northern China. First, all farmers who have to pay to pump water - whether from the ground or from a canal - are in effect paying a volumetric charge for water. Since pumping costs are many times greater than the resource and infrastructure fees, this means that their water charges are already largely volumetric. Therefore, to the extent that the management system permits farmers to consciously plan their use of water, they are already making decisions about the use of water that reflect a comparison of price sand returns. Either such farmers are already using water efficiently or the price mechanism is not working to direct their choice of technique. Secondly, farmers are paying a very high price indeed for their irrigation water. Some farmers in Beidong were paying 10 percent of their gross annual household incomes for water. Although the published prices of water in northern Chinaare RMB0.10 - 0.15/m^3, the effective price paid by these farmers is over RMB3.00/m^3, slightly higher than the price paid by residents of Melbourne for household water (Essential ServicesCommission,2005). These implications of pumping costs seem to have escaped the proponents of market prices for water. Thirdly, since elasticities are solow, any price increase is effectively a tax on farmers for little return in water saved. One forecast, based on data for Shanxi province, estimated that a tenfold increase in the price of surface water would reduce social welfare by 39 percent even though farmers would increasingly turn to groundwater (Fangand Nuppenau, 2004). Since farmers are the poorest group in China, especially in north China, it seems especially unreasonable to tax them in this manner (Ahmad, 2000; Molle, 2001; Perry, 2001). Proponents of market prices for irrigation water thus have to propose income subsidies (that are independent of water) to compensate farmers for such losses.

7.3 Implication

Our evidence and argument prompt two different kinds of conclusions. The first concerns the nature of the claims that are made about the efficiency of irrigated agriculture in northern China; the second reflects more positively on the nature of the solutions to water scarcity that might work and be equitable.

The claims that are made about the use of water by irrigation farmers in northern China include the contentions that the price of water is too low to encourage farmers to be efficient; farmers are not charged volumetric prices and so they are not encouraged to conserve water; water is scarce in large part because farmers are profligate in their use of water; and proper pricing of water will not affect equity. None of these contentions is true. Farmers have to pay not only the official charges for water but also the costs of pumping it onto their fields; these latter charges might be 30 times the official price of water and raise the effective price of water to levels comparable to those in water-scarce developed countries. Once pumping is included, farmers are paying prices that are volumetric: the more water they pump, the more they pay. Furthermore, the inefficiency of farmers arises in large part from the manner in which water is delivered to them: the system offers no rewards for care in the use of water and instead rewards greed. And, finally, although it might be true that higher prices do not affect equity within a village, in fact they would have substantial effects on inter-sectoral equity, with farmers becoming worse off in comparison to urban dwellers. This summary implies that an appropriate means of raising the efficiency with which irrigation water is used must be different from the widely advocated model of universal pricing of individual farmers. Like Haddad (2000a), we do not reject the idea of rights and markets per se but seek to tailor them to fit the physical, institutional, development, and social realities of northern China. First, the fact that farmers who pump are already facing (high) volumetric prices implies that they already are given incentives to irrigate as efficiently as they are able under current institutional arrangements. Thus appropriate systems of management first need to be put into place and only then should additional incentives be given to farmers to become more efficient users of water. The critical criterion for an appropriate system of management is that it provides farmers with the ability to use water efficiently if they choose. And, we have argued, that criterion is met if farmers (or groups of farmers) are allocated water rights, which themselves depend on storage systems that enable farmers to use their water when they need it. If farmers or groups have defined rights to water at times they need it, they have the capacity to use water efficiently - a capacity that they do not now have.

Once such a system of defined rights is established, then the question of price and efficiency can be addressed. The weakness of present proposals for pricing irrigation water is that they intend to charge farmers for use of water without providing them a compensatory income. This is the source of the inequity. The appropriate solution would seem to be to provide farmers with the right to sell their water to urban and other users at market prices. However, allocating these rights is unlikely to work at an individual level, since metering is impracticable and water markets are institutionally incompatible with social practice in China. But it could be done collectively - at the level of an irrigation district, for example: the district sells some of its water entitlement to the large cities of northern China. That income can then be used to pay for the infrastructure that is needed to save the estimated 50 percent of China's surface irrigation water that is lost to seepage and evaporation in irrigation channels. Cities buy the water that they need; farmers are provided with the capital with which to become more efficient users of water; the inequities of high prices are avoided. The markets thus established are local (between cities and nearby irrigation districts) and are at an appropriate scale (the irrigation district). Once such a system is in place, then and only then can the questions of water management at the scale of individual farmers be appropriately addressed.

What ever the design of the system of water transfers from agriculture to urban-industrial users, it is critical that the money transfers occur at the same scale as is the principal source in inefficiency. In agriculture on the north China plain the principal source of in efficiency is the loss of water in irrigation canals.

The provincial bureaus of the YRCC and the irrigation districts within them are the managers of these canals. Money must be transferred to those bureaus and irrigation districts and all of it spent to reduce their seepage losses and raise their ability to provide water when farmers need it. Otherwise, farmers will lose water. If, by contrast, compensation for water transfers is paid directly to farmers, there is no additional money with which the managers of canals can pay to reduce the technical efficiency losses, and agricultural output falls. (This is the fundamental weakness of the market-prices-plus-compensation model.) If compensation is paid to other agencies of government, there is no reason to suppose that it will be applied to reducing efficiency losses in irrigation. Even if a market-like transfer of money for water represents the best deal that farmers can get for what might be regarded as an inevitable loss of their present de facto rights to water, it is still critical that the transfers be designed in the manner we have indicated.