Sustainable Natural Resource Use in Rural China: Recent trends and Policies

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Abstract

In this paper we provide an overview of recent trends in the availability and quality of land and water resources in rural China, and examine the common presumption that rural resources are rapidly degrading in China. Data based on consistent definitions and measurement methods that have recently become available are used to that end. In addition, we analyse the impact of new policy initiatives to introduce market-based instruments and new institutions to address land degradation and water scarcity problems.

We find that the decline in cultivated area has accelerated in the beginning of the new century. Ecological recovery programs, not urbanization and industrialization, are the major factor causing this decline. Ecological recovery programs are also a major force behind the increase in forest land area and the reduction of water erosion. Modest successes can be observed in the protection of wetlands and (until the mid-1980s) for the average quality of cultivated land. On the other hand, degradation of natural grassland and wind erosion have become much more severe in recent decades.

In northern China, particularly in the 3-H (Hai & Luan, Huai and Huang) river basins, the availability of water has tightened. Groundwater tables have fallen considerably in the Hai river basin, because farmers increasingly rely on groundwater for irrigation. Evidence on other parts of northern China is mixed. Pollution of surface water is getting worse since the beginning of the 1990s in two major lakes in southern China and until recently in the rivers in northern China. Water quality problems in the larger rivers in southern China are less severe and getting less. These problems are to a large extent caused by agriculture-based non-point source pollution, especially in the major lakes and reservoirs.

The sloping land conversion program, water pricing, and the establishment of water user associations and payments for environmental services projects are used as cases to examine the introduction of market-based instruments and new institutions. We argue that less government interference in the implementation of these instruments and institutions is likely to enhance ecological as well the economic benefits. Moreover, supportive measures to improve the functioning of land and labor markets are usually needed to ensure the sustainability of the impact of interventions.

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1. Introduction

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 (NDRC, 2006). Investing in rural areas and stimulating consumptive expenditures by rural households is also an important element of China's four trillion yuan stimulus package that was 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 (Mol and Carter, 2006; Xu et al., 2006a; Huang, 2000; Rozelle et al., 1997). However, recognition of the role of more decentralized policy measures, interventions by more informal, non-governmental 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 3 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 paper ends with drawing a number of policy implications. Details about the data sets used for assessing the trends in natural resource use, their accuracy, quality and (in)consistencies can be found in the Appendix.

2. Land availability and quality

Data on the size of the cultivated land and on land degradation processes are plagued with severe measurement problems. Annual data are available on the size of the cultivated land in mainland China, but before the First Agricultural Census held in 1996 the cultivated land area was underreported by more than 30 percent (see Appendix). Until recently, available evidence on land degradation in China consisted mainly of cross-section data for a single point in time, incomplete time series data based on inconsistent definitions and/or measurement methods, and evidence from isolated case studies. This piecemeal evidence did not withhold some authors from drawing far-reaching conclusions about the increasing severity of land degradation processes in China (see e.g. Forestier, 1989; Huang, 2000; World Bank, 2001; Wang, 2004).[1] In recent years, a number of studies have used the available data on cultivated land size to derive consistent estimates before and after 1996. Moreover, new nationwide evidence on different aspects of land degradation, based on data collected for several points in

[1] A more balanced assessment of the available information on the degradation of land other natural resources in rural China by the mid-1990s can be found in Rozelle et al. (1997) That study concludes that successes have been enjoyed in some areas (e.g. deforestation, salinization and water pollution), while problems continue in other areas (e.g. soil erosion, grassland degradation and availability of cultivated land)..

time by the Ministry of Land and Resources of China (MLR) and other institutes, has been published in a number of studies. These data allow us to sketch a more systematic picture of trends in land availability and in land degradation and rehabilitation in mainland China in recent decades, and to assess to what extent land resources in China are heading towards an alleged environmental disaster. In this section we first review changes in the size of cultivated land, and the factors responsible for such changes. Next, we summarize the results of recent studies on land degradation processes, distinguishing between different degradation processes and methodologies for measuring degradation.

Availability of land

With a territory of 960 million hectare, China's cropland area in 2007 was estimated at 121.7 million hectares of cultivated land, plus 11.8 million hectares of garden land, 236.1 hectares of forest land, 261.8 hectares of pastureland and 25.5 million hectares of land for other agricultural purposes (MLR, 2010). With an average of 11 persons per hectare, cultivated land in China is relatively scarce as compared to the global average of three persons per hectare of cropland in the 1990s (Ramankutty et al., 2002). Yet, several countries have even less cultivated land available for agricultural production. In Indonesia, Vietnam, Bangladesh, Egypt, South Korea and Japan, cultivated land availability in 1999 ranged from 12 – 28 persons per hectare (Hansen, 2002).

Lindert (1999: 720-723) examines trends in the size of the cultivated land area as one of the dimensions of China's soil endowment problem. Correcting for underreporting in official statistics, the study finds that the cultivated land area declined from 137 million hectares in 1978 to 131 million hectares in 1993.[2] Combining Lindert's (1999) estimate of cultivated land with recent data from the MLR, shows that a total size of 15.3 million hectares, or 11 percent of the total cultivated land area, has been lost during the period 1978-2007. At a global scale, such a rapid declines is quite exceptional. During the 20th century, only Japan between 1960 and 1990 witnessed a more rapid decline in cropland area (from 19.9 to 16.1 percent of its cultivated land area, i.e. a 19 percent decline) over a similar period (Ramankutty et al., 2002).

Land monitoring data of the MLR for the period 1987-2000 analysed in Fisher et al. (2007) show that annually 0.7 million hectares of farmland have been lost and 0.4 million hectares have been reclaimed on average, resulting in an annual net loss of 0.3 million hectares, about 0.25 percent of the cultivable land resource.[3] Driving forces behind the loss of land are the conversion of sloped farmland in fragile and low-productivity environments into forests,

[2] These estimates are consistent with those obtained by Heilig (1999). Using data on land use changes provided by the Chinese State Land Administration, and combining these data with the stock at the end of 1995, Heilig derives that the cultivated land size has declined from 132.5 million hectares in 1988 to 131.1 million hectares in 1995.

[3] Deng et al. (2006), on the other hand, conclude from Landsat satellite images that the conversion of cultivated land to other uses was surprisingly low during the period 1986-2000. Their analysis shows that the annual increase in cultivated land in fact exceeded the loss of cultivated land during that period, resulting in an annual increase of cultivated land of 0.18 million hectares (see also Appendix).

shrub or grassland (38 percent of the lost land), (urban) construction activities that often tend to take away high-quality farmland (22 percent), transformation of cropland into orchards and fishponds (25 percent), and severe damage caused by natural disasters (15 percent). Due to the strong improvements of crop yields per hectare of about 1 percent per year, however, the domestic availability of rice, wheat and maize has not suffered from the area decline (Fisher et al., 2007).

[Table 1]

Using the land monitoring data of the MLR, Tan et al. (2007) have made a more detailed examination of the sources of cultivated land loss between 1989 and 2005. Their study finds that the annual land loss shows a marked acceleration towards 1.2 million hectares since 2000. Ecological restoration programs, particularly the Sloping Land Conversion Program (SLCP), were the main cause of this acceleration. These programs contributed 67 percent to the 9.2 million hectares decline in cultivated land over the period 2000-2005, while land loss to urbanization was responsible for 14 percent of the decline (see Table 1). These results confirm preliminary findings for the period 2000–2003 presented in Deng et al. (2006).

As in many other countries, land conversion into urban construction land takes place in areas with relatively high agricultural productivity. In China, rural-urban land conversion is concentrated in relatively flat, high precipitation land in the south and east. Ecological restoration programs, on the other hand, take place in low productivity areas, particularly in western China, with yields less than 30% of the national average. As a consequence, the average productivity of land taken out of cultivation was higher than the national average before the large-scale implementation of the SLCP at the end of the 1990s, but was significantly lower since then (Deng et al., 2006; Tan et al., 2007).

Land degradation

In a detailed study of land resource degradation, Zhang et al. (2006) define soil degradation as the transition of land use types into types with lower ecological service functions or productive capacity. Their study uses annual data from the land resource surveys conducted by the MLR to assess the relative importance of seven major land degradation processes during the period 1991-2002. The processes are sandy and rocky desertification, secondary salinisation, non-agricultural occupation, deforestation, natural grassland degradation, and wetland shrinkage; Table 2 gives an overview. Taking the seven processes together, land resource degradation in China is estimated to amount to 1.6 million hectares annually, rehabilitation to 1.1 million hectares, so that an estimated net annual resource loss of 0.5 million hectares results (or 6.44 million hectares during the entire 1991-2002 period). The main sources of resource degradation during the period 1991-2002 were grassland degradation and conversion to non-agricultural uses, while afforestation and restoration of sandy deserts contributed most to rehabilitation.

[Table 2]

Natural grassland covers 27.2 percent of the total available land area of 960 million hectares.[4] During the period 1991–2002, an area of 8.06 million hectares (that is, 3.1 percent of the available natural grassland) was lost to unused land or transformed into other land use types, mainly woodland, man-made grassland and cultivated land. During the same period 1.68 million hectares of land was rehabilitated into natural grassland, resulting in a net decline of 6.38 million hectares (equal to 2.4 percent of the available natural grassland).

The second major land degradation problem in the analysis by Zhang et al. (2006) is non-agricultural land use (for residence, industries and mining, and traffic). During the period 1991-2002, 4.97 million hectares were converted into built-up land. The largest share of this land, 2.95 million hectares or 59.3 percent, was used before as cultivated land and was often located in high-productivity areas. Converting non-agricultural land into other land types amounted to 0.86 million hectares, giving a net non-agricultural land occupation of 4.11 million hectares (equal to 0.35 million hectares per year).

A third degradation problem that worsened during the period 1991-2002 is that of secondary salinisation. Despite government campaigns to reclaim saline land, the size of the saline-alkali land increased by 0.15 million hectares on balance. Pasture land accounts for about three-quarters of the increase in this type of land degradation (Zhang et al., 2006: Figure 2).

The other four transition processes examined by Zhang et al. (2006) all show signs of improvement. Forest land in China covers around 235 million hectares, that is 24.5 percent of the total land area. The deforested area between 1991 and 2002 equalled 1.72 million hectares. But the area of afforestation and plantation increased during the same period by 3.43 million hectares, mainly as a result of large-scale government programmes such as the SLCP, the Natural Forest Protection Program (NFPP) and the Three North Shelterbelt Construction Program (3NSCP). The result was a net increase of 1.72 million hectares in forest land.[5]

Sandy desertification, defined by Zhang et al. (2006) as the conversion of land into sand, bare and waste land, affected 3.01 million hectares. Degraded grassland accounted for the largest share (58 percent) of sandy desertification (Zhang et al., 2006: Figure 2). On the other hand, large-scale desertification control programs, particularly the 3NSCP in the agropastoral zone of northern China (Zhang et al., 2007), have rehabilitated a total of 4.30 million hectares of sandy area. On balance, the total sandy desert land area therefore decreased by 1.29 million

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[4] Besides natural grassland, the total grassland area in China also comprises improved grassland and man-made grassland. The total area covered by grassland in China equals 400 million hectares or 41.7 percent of China's total territory (NBS, 2010).

[5] Data collected during the national forest censuses show that the net increase in forest area between 1977-81 and 1988-93 was as large as 18.4 million hectares. This increase reflects the major reforestation efforts of the 1980s and the beginning of the 1990s, and was concentrated in particular in the East and South (World Bank, 2001: Table 2.6).

hectares between 1991 and 2002. Similar trends, but on a smaller scale, can be observed for rocky desertification.[6] Together, both types of deserts make up 207 million hectares of land, i.e. 21.6 percent of China's total territory. Net rehabilitation therefore reduced the size of China's deserts by 0.7 percent during the period 1991-2002.

The shrinkage of wetlands, mainly due to the reclamation for cropland and non-agricultural land, was more than compensated by public integrative control and sustainable rehabilitation programs, resulting in a 0.92 million hectares increase in the total area of wetlands in mainland China.

The study by Zhang et al. (2006) provides useful information on major trends in land conversion from one type to another, but it does not provide insights into quality changes of land use types that remain unchanged. For example, natural grassland may increasingly suffer from overgrazing but still remain classified as grassland in a land resource survey, and will therefore not be considered grassland degradation in the analysis. Studies by the Ministry of Agriculture cited in World Bank (2001), Li (2004) and Nan (2005) indicate that around 90 percent of the usable grassland in China is degraded to some degree while 34 percent suffers from moderate or severe degradation. The total area of degraded grassland has increased by 95 percent between 1989 and 1998, with a notable acceleration after 1993 (SEPA, 1998 cited in World Bank, 2001). The quality of the forested land has also declined in recent decades. World Bank (2001) shows that, despite the net increase in forested area since the mid-1970s, the stock of mature and over-mature trees has declined by about 40 percent, and its share in the standing stock by about 20 percent between the mid-1970s and the beginning of the 1990s.

[Table 3]

The results obtained by Zhang et al. (2006) also fail to give any insights into the severity of soil erosion and its trends over time. Due to its geography and climate, land in China is prone to soil erosion (World Bank, 2001). Trends in land areas affected by water and wind erosion since the mid-1980s are examined by Heerink et al. (2009) and Heerink et al. (2010), using data collected in the three national soil erosion surveys that have been held so far. The findings are summarized in Table 3. The land area affected by erosion has declined from 38.5 percent in the mid 1980s to 37.5 percent in the mid 1980s and has remained almost stable since then. The area affected by wind erosion is larger than that affected by water erosion (21 vs. 17 percent in 2001/02). Moreover, wind erosion area has increased during both periods, while the area affected by water erosion has declined, particularly between the mid-1980s and mid-1990s. This finding contradicts the increase in water erosion area from 13.7 in 1986 to

6 In another publication based on the same methodology, however, Zhang et al. (2007: Table II) report that the area of desert land decreased by only 0.23 million hectares between 1991 and 2001.

19.0 percent in 1996, based on the annual surveys undertaken by the MWR, reported in Huang (2000) and World Bank (2001).[7]

In Western China, not only the area affected by wind erosion increased between the mid-1980s and the mid-1990s but also the area affected by water erosion (see Table 3). During the second period, however, the area affected by water erosion declined by 0.2 and 0.5 percentage points respectively in Northwest and Southwest China. Particularly the areas affected by water erosion in Sichuan, Shaanxi and Yunnan Provinces declined considerably during this period (that is, between 1.1 and 2.4 percentage points).[8] Liu et al. (2008) likewise note that water erosion in Western China declined significantly since the start of the SLCP and NFPP in 1998-1999. The area affected by wind erosion is to a large extent located in Northwest China, and increased considerably during both periods: 1.3 percentage points form 1985/86 to 1995/96 and 1.0 percentage points between 1995/96 and 2001/02 (see Table 3). During the latter period, the increase in wind erosion was concentrated mainly in Inner-Mongolia and Qinghai, where the areas affected by it increased by 2.4 and 3.1 percentage points, respectively.

Table 3 further shows that especially the area affected by the three most severe categories of water erosion (intense, more intense and severe) declined between the mid-1980s and mid-1990s. At the same time there was a major shift between the different wind erosion classes, from the relatively mild erosion classes (light and medium) to the most serious erosion classes (more intense and severe). The increase in erosion area between the mid-1990s and the beginning of the 21th century was caused in particular by an increase in the land area belonging to the light and medium wind erosion classes.

Cao (2008) argues that inappropriate design of the 3NSCP and the SLCP is a major cause of the increase in wind erosion in arid and semi-arid China. Due to the emphasis on planting commercial trees in such programs, soil moisture content is declining and sunlight under tree canopies is reduced, resulting in decreased vegetation and low tree survival rates on afforested plots. What is needed instead is the recreation of natural ecosystems (such as natural steppe and grassland vegetation) that take into account the carrying capacity of such systems.

Huang (2000) compares soil erosion in China with other countries and finds that China is one of the most serious eroded countries in the world. The share of eroded land in the total land area in China is more than three times the average for the world. It is about 2.4 times higher than in the rest of Asia, almost three times higher than in Africa, and 7-8 times higher than in North-America and Oceania. Especially wind erosion is a relatively severe problem in China. The share of land affected by wind erosion in China is 6.5 times the world average (Huang, 2000: Table 3).

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[7] See Appendix for an assessment of the quality and comparability of soil erosion data from different sources.
[8] Provincial soil erosion survey data on changes in erosion areas are available upon request from the authors.

The data on land degradation trends presented in the aforementioned studies refer to the total land area, cultivated land as well as grassland, forest land and other land.[9] For examining potential limitations to crop production, degradation processes on cultivated land are more relevant.[10] Changes in the quality of the cultivated land have been examined by Lindert (1999), using soil samples collected in the 1930s, 1950s and 1980s. He finds that soil organic matter and nitrogen have declined between the 1950s and 1980s in most regions, but not in the (water erosion-prone) Southwestern rice area. By contrast, other topsoil characteristics (phosphorous, potassium, acidity / alkalinity) showed favorable or neutral trends. Using the marginal productivity of each characteristic as weights, the study finds that the average quality of China's cultivated soils increased by 4 to 8 percent between the 1950s and 1980s.

A more recent cause of concern is the increase in soil contamination with heavy metals. Wastewater irrigation zones are spreading in China and have become a major reason for soil contamination, accounting for about 4 million hectares of agricultural land. The harvested produce is likely to contain heavy metals such as mercury, cadmium, lead, copper, chromium, and arsenic. Estimates are that roughly 10 million hectares of arable land is now polluted in China, of which 2.2 million hectares were irrigated by polluted water with heavy metals as one of the major pollutants (SEPA, 2006). Increasing water scarcity is considered the main reason for using polluted wastewater in irrigation.

Main findings

Summarizing these trends in land availability and quality, we can conclude that the size of the cultivated land has steadily declined since the end of the 1970s, with an acceleration at the beginning of the 21st century caused by ecological recovery programs like the SLCP. Such ecological recovery programs have been successful in increasing the size of the forest land and in reducing water erosion, but have probably contributed to an increase in the area affected by wind erosion. Degradation of natural grassland has become much more severe in recent decades, while secondary salinization also increased. The average quality of cultivated land has improved over time at least until the 1980s, and (due to the SLCP) also in the beginning of the new century.

3. 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 2,812.4 billion m^3 until the year 1999,

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9 An exception is Yang (1994) cited by Huang (2000). Using data from the second national soil erosion survey, Yang (1994) finds that 34.3 percent of the cultivated land suffers from erosion. To our knowledge, analyses of time trends in erosion on cultivated land are currently not available.

10 It may be noted, however, that studies that use erosion indicators for total land as proxies for the erosion of cultivated land have found a significant negative correlation of such erosion indicators with agricultural production in China (see Sinden and Xu, 2003 and Heerink et al., 2009).

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).[11] 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 2,849 m^3 per person per year in 1980 to 1,785 m^3 in 2009 (NBS, 2010), and is rapidly approaching the internationally accepted thresholds for defining water stress (1,700 m^3 per person per year) and water scarcity (1,000 m^3 per person per year). Although per capita water availability is slightly higher than in India, it is only one-third 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 (see Table 4). Water availability in the North (757 m^3 per person in 2003) is almost 25 percent below the water scarcity threshold, while water availability in the South (3,208 m^3 per person) is relatively abundant. Large differences also exist within the northern region, with the so-called 3-H river basins - the Hai & Luan, Huai and Huang (= Yellow) river basins - facing the most severe water scarcity; per capita water availability in the 3-H basins was estimated at 499 m^3 in 1999 (World Bank, 2001; World Bank et al., 2001).

[Table 4]

Due to the growing scarcity of surface water, groundwater use in agriculture is rapidly increasing. The number of tubewells used for groundwater irrigation has increased from 0.2 million in 1963 to 4.7 million in 2003 (Zhang et al., 2008) and to 5.2 million in 2007 (MWR, 2009). Nearly all these tubewells (95 percent) are in northern China, even though only 30

[11] In 2008, the State Environmental Protection Agency (SEPA) was transformed into the MEP.

percent of the groundwater resources are located in the northern part of the country (Wang et al., 2007). Using 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 percent of the 448 villages that were surveyed. In 35 percent of the villages the groundwater level showed little or no decline since the mid-1990s, while in 48 percent of the villages the water table had declined. In 8 percent of the villages, the rate of decline exceeded 1.5 meters 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 percent of the monitoring sites while the level increased in the remaining 39 percent. The total groundwater volume increased in two out of eight years (2003 and 2005; years in which precipitation was relatively high), while it declined in the other six 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 meters 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: Figure 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 cubic meters of non-rechargeable deep-aquifer groundwater were mined in China in 2000, mainly for agricultural purposes (World Bank and SEPA, 2007).

[Figure 1]

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 m^3 in 2008 (see Figure 1). 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 percent over the period 1997-2008. By using more efficient irrigation systems and cultivation methods, total water use in agriculture declined by 12.4 percent from 392.0 to 343.3 billion m^3 between 1997 and 2003 despite an increase in the irrigation area by 5.4 percent during this period (World Bank, 2006). After 2003, however, water use in agriculture increased by 6.7 percent to 366.3 billion m^3 in 2008, while the irrigation area increased by 8.3 percent during the same period (NBS, 2003-2009). As a share of total water use, the use of water in agriculture has steadily declined from around 80 percent in 1980 to 62.0 percent in 2008 (World Bank, 2006; NBS, 2010).

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 (World Bank, 2001, 2006; MEP, various years). 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 percent of the monitoring sites in the North is suitable for human consumption after treatment as compared to 85 percent in the South (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 run-off 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 (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 percent of the nutrient inflow into the Chao Lake and the Dianchi Lake is derived from agricultural runoff (Shalizi 2006: p.11).

[Table 5]

In around 50 percent 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 of major 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 km^2 (World Bank et al., 2001).

Main findings

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 China. Agricultural runoff is the main source of pollution in these lakes and also a major source of the pollution of river waters.

4. Policy initiatives to reverse natural resource degradation

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.

The Sloping Land Conversion Program

The start of the NFPP in 1998 and the SLCP in 1999 marked the implementation of two of the world largest projects to protect and expand forest and grassland resources. The severe drought of 1997 and massive floods in 1998 undoubtedly hastened the decision by the central government to initiate such major interventions. The SLCP (also known as the Grain-for-Green Program or the Grain-to-Green Program) complements the NFPP, with the latter aimed at protecting and restoring natural forests through such means as logging bans, and the former aimed at addressing another major cause of erosion i.e. farming on steep slopes (Liu et al., 2008).

The SLCP aims to convert 14.7 million hectares of cropland on steep slopes in the upper reaches of the Yellow and Yangtze River Basins back to forestland and into natural grassland by 2010. The criterion used is that the slope should be at least 15 degrees in northwestern China and at least 25 degrees elsewhere. Another 17.3 million hectares of vegetative cover will be created by afforesting barren land (Liu et al., 2008). Two associated SLCP goals are to alleviate poverty and to assist farmers to shift to more sustainable structures of production (SFA, 2003).

The program generally uses a top-down approach in the selection of villages to be included in land retirement, with retirement quotas distributed by higher over lower level governments (Zuo, 2001). In addition, it uses a public payments scheme that directly engages rural households as core agents of project implementation on the (stated) principle of volunteerism; in this respect, it differs from other large-scale ecological recovery programs in China, such as the 3NSCP and the NFPP (Bennett, 2008). The government offers farmers 2,250 kg of grain or 3,150 yuan (around 450 US dollar) per hectare of converted land per year in the upper reach of the Yangtze River Basin, and 1,500 kg of grain or 2,100 yuan (around 300 US dollar) per converted hectare per year in the upper and middle reaches of the Yellow River Basin (Feng et al., 2005; Bennett, 2008).[12]

The differences in subsidy between the two regions appear to reflect differences in opportunity costs for participation of farm households between the regions (Bennett, 2008). The duration of the subsidy depends on the outcome of the conversion: 2 years for grassland, 5 years for economic forests (fruit trees, trees with medicinal value) and 8 years for ecological forests (trees with timber value, such as pine trees and black locust). Income earned from trees and grassland planted under the program is exempted from tax (Bennett, 2008). The

[12] In addition, the government offers 300 yuan (around 43 US dollars) per hectare for miscellaneous expenses and free seeds or seedlings (worth around 750 yuan, or 107 US dollars, per hectare) in cash or in kind.

grain oversupply at the end of the 1990s and the rapidly increasing financial resources of the Chinese government provided a stable foundation for implementing the SLCP (Tao et al., 2004).

By the end of 2006, almost 9 million hectares had been converted into forest or grassland under the SLCP.[13] A total of 32 million rural households were involved in these activities. At that time, after 8 years of implementation, the central government had invested a total amount of 130 billion yuan (around 19 billion US dollar) in the SLCP (WDO & DRC, 2007). The planned total investment in the SLCP is 220 billion yuan (around 32 billion US dollars) by 2010 (Liu et al., 2008).

Evidence presented by Liu et al. (2008) shows that the ecological effects of the SLCP are generally positive. Surface runoff and soil erosion are significantly reduced on converted plots. Moreover, the program improves soil fertility and physical properties of soil structure of converted plots and contributes to saving water resources and reducing desertification. On the other hand, the diversity of planted tree species in the program tends to be low. In most regions, converted land is planted with a single or just a few tree species. Moreover, the survival rates of planted trees is lower that the rate stipulated for subsidy delivery in many regions, due to a lack of technical support to farmers, the absence of secure land tenure, limited access to off-farm employment, and other factors (Weyerhaeuser et al., 2005; Bennett, 2008; Cao, 2008; Bennett et al., this volume).

The emphasis on tree planting in (semi-)arid areas such as the Loess Plateau region is questionable, as the water used by such trees puts a high demand on the region's strained water resources (Bennett, 2008; Cao, 2008). The CCICED (2002) points out that the implementation of the SLCP has not been tailored to local conditions, that the SLCP puts too much emphasis on tree planting, and that it does not give sufficient consideration to the ecological and economic functions of grasslands in semi-arid areas and the need to restore the original vegetation of these ecosystems. In addition, the targeting of sloping and marginal lands in the program has been less than optimal. Xu et al. (2004) and Uchida et al. (2005) find that relatively productive land on low slopes was retired in some regions.

Closer examination of the funding mechanisms of the SLCP shows that the bulk of the financing comes from the central government. The latter provides partial subsidies to local governments, but the cost of monitoring, grain transportation and other implementation activities should be covered by the local governments themselves (Liu and Zhou, 2005; Bennett, 2008). In practice, the counterpart contributions of local authorities in terms of funding and labor input have remained modest (Xu et al., 2006b). Available evidence on the compensations received by farm households shows significant shortfalls in the amount of subsidies actually reaching the farmers (Xu and Cao, 2001; Zuo, 2001; Bennett, 2008). These shortfalls are caused in many cases by local government efforts to recoup implementation

[13] Moreover, 11.7 million hectares of barren land had been afforested by the end of 2006 (Liu et al., 2008).

costs, and in other cases by the administrative burden and backlog in inspection and certification (Bennett, 2008).

Participation in the SLCP is commonly believed to be quasi-voluntary, with households being strongly encouraged by local governments to participate. Results from a survey held in 2003 among participating and non-participating households in six counties in three provinces[14] shows that 62 percent of the participating farm households and 26 percent of the non-participating farm households felt they could choose whether or not to participate, and only 30 percent had the autonomy to decide which plots to retire (Bennett, 2008; Xu et al. 2010). This clear deviation from the program's stated objective of volunteerism means that a large share of the efficiency gains that can be gained from implementing a PES scheme instead of using a command-and-control approach are not realized. Introduction of more market-based mechanisms, such as auctions for land retirement, to better align the costs of the environmental service provider with the benefits of the buyer (the Chinese government in this case) could help to reduce overall program costs and involuntary participation of farm households in the program (Bennett, 2008).

The program has been moderately successful in alleviating poverty. Results from the aforementioned survey held in 2003 reported in Xu et al. (2004) and Uchida et al. (2007) shows that, even though most households may not have participated voluntarily and actual payments to farm households may have been lower than the promised amounts in some areas, the program brought some modest positive effects. It is implemented mostly in regions that are fairly poor, but did not target the poorer households in particular. The program did not have a significant effect on the income of participating households as compared to non-participating households, but it has induced participating households to shift their activities from grain production (in many cases on degrading soils) towards livestock production and investing in housing and other assets.

The program was not successful in shifting labor into off-farm sectors during the first few years of implementation (Xu et al., 2004; Uchida et al, 2007), but significant higher shifts to off-farm employment have been found among participants than among non-participants after five program years (Uchida et al., 2009). The shift towards off-farm employment was highest among liquidity-constrained households and among younger adult family members with higher educational attainment. Groom et al. (2010) and Grosjean and Kontoleon (2009) argue that even larger shifts towards off-farm employment may occur when market and institutional constraints are addressed (primarily weak and incomplete land property rights and high labor mobility transaction costs). Removing such constraints goes at the root of oversupply of farm labor and thereby stimulates the long-term viability of the SLCP, that is the sustainability of its impact after the program ends.

[14] The survey covers 359 households from six counties in Shaanxi, Gansu and Sichuan provinces, with 75 percent of the households participating in the SLCP.

Water scarcity management

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 and 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 percent 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 percent nationwide (MWR, 2002a,b; Liao et al., 2008).

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., 2007, 2008). 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 et al. (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 percent 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 percent of interviewed villages knew groundwater markets, but by 2004 there were groundwater markets in 44 percent of the

[15] The data covered 68 randomly selected villages in Hebei, Henan, Shanxi and Shaanxi provinces.

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 et al., 2004, Wang et al., 2005). By 2004, as much as 70 percent 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 (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 and 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 a Water 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 (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 et al. 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 and 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

[16] Other major elements of the pilot project are the development of the industrial and services sector to reduce the pressure on the agricultural sector, and construction of an engineering system that optimizes the water distribution (particularly construction of a water-saving irrigation system and installation of meters for water users).

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 and 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 (Zheng and Zhang, 2006; Guo, 2007). 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 percent 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; Xie 2007; Wang et al., 2010).

An empirical study among WUAs in Ningxia, Gansu, Hubei and Hunan Provinces by Wang et al. (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 percent 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.

Main findings

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.

5. Conclusion

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 percent and attaining a capability of producing 540 million tons of grains by the year 2020 (NDRC, 2008). During the period 2007-2009, the average annual grain production stood at 520 million tons (with a self-sufficiency ratio of 97 percent). Hence, a production increase of around 4 percent 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 percent of the land suffers from (water and wind) erosion, 90 percent of the usable grassland in China is degraded to some degree, and water in 80 percent of the major lakes and reservoirs and 45 percent 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 (MLR, 2007 & 2010) and ended in 2009 before the program had fully reached its intended scale. Yet, ongoing urbanisation 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 above; 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. Results from an analysis of China's urban expansion during the period 1995-2000 by Deng et al. (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., 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; 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 percent 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 et al., 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; Ngaido and McCarthy, 2005; Nelson, 2006).

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 run-off 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., this volume).

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 percent 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.

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Appendix: Data sources and quality

Land availability

Data about the availability of cultivated land come from the land use change monitoring system of the Ministry of Land and Resources (MLR 1998-2000, 2001-2005, 2007, 2010). Until 1996, the bottom-up bureaucratic administration system was used for reporting cultivated land areas. As local officials had an interest in inflating agricultural yields and avoiding agricultural tax payments, this resulted in an underestimation of the actual cultivated land size (Hansen et al. 2002). The first modern agricultural census that was held in China in 1997 provided the first accurate benchmark data on cultivated land. Estimates of the cultivated land area in 1996 went up from 95.0 million hectares to 130.1 million hectares (Gale 2002, Hansen et al. 2002, MLR 1998-2000). Since then, cultivated land area has declined to 121.8 million hectares in 2006 according to the MLR land use change monitoring system. This number is consistent with the size of the cultivated land area in 2006 that was reported druing the second agricultural census that was held in 2007 (NBS, 2008).

Deng et al. (2006) use Landsat satellite images for estimating the conversion of cultivated land to other uses during the period 1986-2000. Their data show surprisingly low conversion rates as compared to the MLR land use change monitoring data. The paper notes that the conversion rates also differ fundamentally from land conversion trends observed in other countries undergoing rapid economic development, but no explanation for the potential causes of these discrepancies is given. For our analysis of recent trends in cultivated land area we therefore prefer to use the MLR land use change and agricultural census data.

Land degradation

Data presented in Table 2 on land degradation during the period 1991-2002 are also based on the land use change monitoring system of the MLR. Using annual data on 8 land use types and 46 sub-types, estimates of land degradation are obtained by calculating the transition of land use types into types with lower ecological service functions or productive capacity (Zhang et al., 20006).

Heerink et al. (2009) and Heerink et al. (2010) use data collected in the three national soil erosion surveys that have been held in 1985–86, 1995–96 and 2001–02 to assess trends in soil erosion since the mid-1990s. The national erosion surveys combine Landsat satellite images with information from land use maps (see Table A1 for details). The classification into soil erosion intensity categories is based on annual soil erosion rates (tonnes per km) and soil erosion thickness (mm); see Li et al. (2006) for details. Data obtained in the first soil erosion survey are based on images with a lower resolution and maps with a larger scale than those used in the second and third survey (see Table A1). Moreover, some highly implausible changes in erosion intensity between the first and second survey can be observed for a few provinces in West China (see Li et al., 2006 for details). Observed trends between the second and third survey seem therefore more reliable than observed trends between the first and second survey.

[Table A1]

The national soil erosion surveys are assumed to give more precise erosion estimates than the annual surveys undertaken by the Ministry of Water Resources (MWR). Water erosion data obtained by the MWR, as reported in Huang (2000), show sudden jumps of 2.7 percentage points from 1990 to 1991 and 2.0 percentage points from 1995 to 1996, which suggest that changes in definitions or measurement methods have taken place over the years.

The land degradation trends presented in the aforementioned studies refer to the total land area, cultivated as well as uncultivated, in China. Changes in the quality of the cultivated land (which comprises only 13 percent of the total land area in China) are examined by Lindert (1999), using soil samples collected in the 1930s, 1950s and 1980s. The samples provide information on long-term trends in five major soil-chemistry indicators - acidity or alkalinity (pH), soil organic matter (OM), total nitrogen (N), total phosphorous (P) and total potassium (K). To our knowledge, no attempts have been made so far to update these trends with information from soil samples taken in the 1990s or the new century.

Water availability

Information about water availability in China and its use by agriculture and other sectors is collected by the MWR and published in NBS, *China Statistical Yearbook*. Available water resources are derived from the surface water and groundwater resource, and corrected for double counting.

Information on groundwater levels in the northern China plain is collected since 2001 on monitoring sites located in plain areas of 17 provinces[17] by the MWR (MWR, various years). Additional information has been obtained by Wang et al. (2007) through a regionally representative survey held in 448 villages in six provinces in Northern China in 1995 and 2004.

Water pollution

Water quality in the nine major rivers is monitored through national and provincial water monitoring centers in almost 500 monitoring stations (World Bank, 2006). About 30 pollutants are included in the monitoring schemes, with the worst individual monitored pollutant establishing the water quality grade. Water in each river section is classified into six

[17] Details about the monitoring sites can be found in Table 2 of MWR (various years).

different grades: Grades I–III refer to water that is safe for human consumption after treatment; grades IV–V refer to water that is safe for industrial and irrigation use; and grade V+ refers to water that is unsafe for any use. The percentages of the sections in each watershed that meet these grades are reported in recent years in MEP's annual Report on the State of the Environment in China (MEP, various years). Results from this water pollution monitoring system dating back to 1991 are presented in World Bank (2001, 2006) and Xie (2009). A similar water quality monitoring scheme is in place in China's major lakes and reservoirs, with results published in MEP (various years) and, for total N and total P pollution during 1992-1998, in World Bank (2001).

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Table 1: Changes in cultivated land area, 1989-2005 (million hectares)

	1989-1999	2000-2005	Whole period
Land taken into cultivation:	6.20	2.04	8.24
Land taken out of cultivation	6.69	9.16	15.85
Construction	2.14	1.26	3.40
Ecological restoration	2.53	6.14	8.67
Destroyed by natural disasters	1.19	0.31	1.50
Agricultural structural adjustment	0.83	1.43	2.26
Net land taken out of cultivation	0.49	7.12	7.61

Source: Based on Tan et al. (2007)

Table 2: Land degradation between 1991 and 2002

Land type	**Stock in 2002 (million ha)**	**Changes 1991-2002 (million ha)**		
		Loss from degradation	**Gain from rehabilitation**	**Net rehabilitation**
Sandy deserts	104	3.01	4.30	1.29
Rocky deserts	103	0.33	0.59	0.26
Saline-alkali land	11	0.45	0.30	-0.15
Non-agricultural land	31*	4.97	0.86	-4.11
Forest land	235*	1.72	3.43	1.72
Natural grassland	261	8.06	1.68	-6.38
Wetlands	35	1.13	2.05	0.92
Total		19.66	13.23	-6.44

Source: Based on Zhang et al. (2006);*: Ministry of Land and Resources (2003)
Loss (gain) is defined as the transition from a higher (lower) to a lower (higher) ecological service functions or productive capacity land use type.
Note: Stock of forest land in 2002 also comprises non-forest woodland, such as shrubbery land. In 1996, the non-forest woodland constituted 29.5 percent of the total woodland (Li, 2000).

Table 3: Areas affected by erosion

	Total erosion area			Light erosion	Medium erosion	Intense erosion	More intense erosion	Severe erosion
	Whole country	NW China	SW China					
	Percentage of total land area							
Total erosion area								
1985/86	38.5	55.7	20.6	19.5	8.2	5.0	2.7	3.2
1995/96	37.5	57.4	21.3	17.0	8.5	4.5	3.5	4.0
2001/02	37.6	58.3	21.0	17.2	8.5	4.5	3.4	4.0
Water erosion area								
1985/86	18.8	13.4	18.6	9.6	5.2	2.6	1.0	0.4
1995/96	17.3	13.9	19.2	8.7	5.8	1.9	0.6	0.3
2001/02	16.9	13.7	18.7	8.7	5.6	1.8	0.6	0.2
Wind erosion area								
1985/86	19.7	42.3	2.0	9.9	2.9	2.4	1.7	2.7
1995/96	20.1	43.6	2.2	8.3	2.7	2.6	2.9	3.7
2001/02	20.7	44.6	2.3	8.5	3.0	2.6	2.8	3.7

Source: Heerink et al. (2009), Heerink et al. (2010)

Note: See Li et al. (2006) for definitions of light, medium, intense, more intense and severe erosion.

Table 4: Water availability per capita, 1980 – 2003 (cubic meters)

	1980	**1993**	**2003**
Total	2,849	2,373	2,180
North	964	838	757
South	4,176	3,665	3,208

Source: Shalizi (2006)

Table 5: Annual average water pollution in three major lakes, 1992 – 2006 (mg/litre)

	1992	**1998**	**2006**
Tai Lake			
Total Phosphorous	0.08	0.07	0.08
Total Nitrogen	2.98	1.84	3.17
Chao Lake			
Total Phosphorous	0.25	0.36	0.15
Total Nitrogen	3.83	2.88	1.61
Dianchi Lake[1]			
Total Phosphorous	0.23	0.41	1.42
Total Nitrogen	2.14	4.76	13.7

Source: World Bank (2001) and SEPA (2007)

[1] : Results for 2006 refer to the outer lake only.

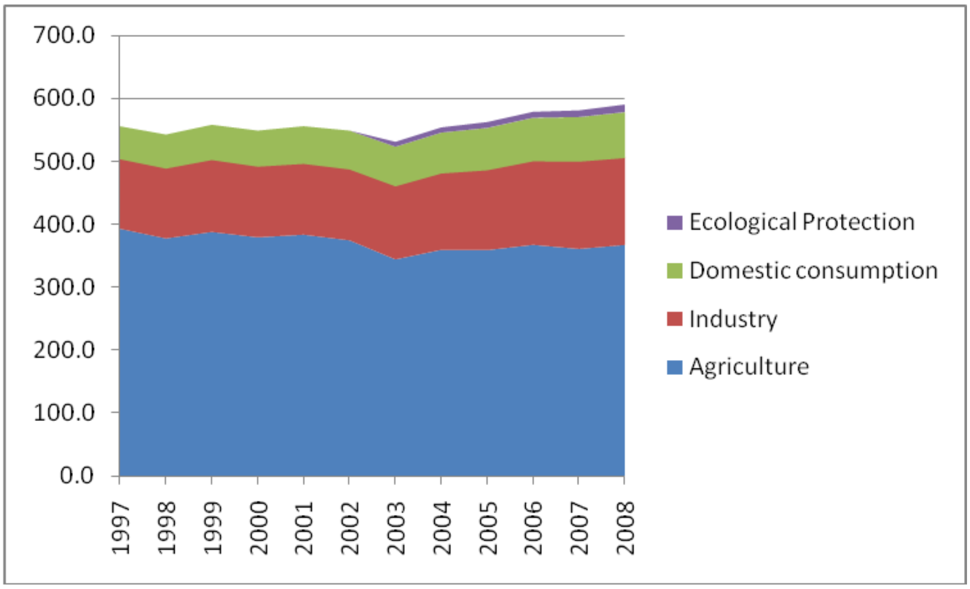
The thresholds beyond which eutrophication becomes a visible problem are Total P > 0.05 and Total N > 0.5 (World Bank, 2001).

Table A1: National soil erosion surveys

	Time of surveying	Time of main data	Type of data	Resolution of data	Supplementary data	Scale of base map
1	1984~1990	1985-1986	Landsat-MSS	80m	Provincial thematic maps	1:500,000
2	1999~2000	1995-1996	Landsat-TM	30m	Digital land use map (1:100,000) of years 1995~1996	1:100,000
3	2002	2001-2002	Landsat-TM	30m	Digital land use map (1:100,000) of year 2000	1:100,000

Source: Li et al. (2006)

Figure 1: Water use by sector, 1997-2008 (in billion m^3)



Source: NBS (various years)