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# Effects of Land Use Change on Soil Carbon Storage and Water Consumption in an Oasis-Desert Ecotone

**Yihe Lü · Zhimin Ma · Zhijiang Zhao · Feixiang Sun · Bojie Fu**

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**Abstract** Land use and ecosystem services need to be assessed simultaneously to better understand the relevant factors in sustainable land management. This paper analyzed land use changes in the middle reach of the arid Heihe River Basin in northwest China over the last two decades and their impacts on water resources and soil organic carbon (SOC) storage. The results indicated that from 1986 to 2007: (1) cropland and human settlements expanded by 45.0 and 17.6 %, respectively, at the expense of 70.1, 35.7, and 4.1 % shrinkage on woodland, grassland, and semi-shrubby desert; (2) irrigation water use was dominant and increased (with fluctuations) at an average rate of 8.2 %, while basic human water consumption increased monotonically over a longer period from 1981 to 2011 at a rate of 58 %; and (3) cropland expansion or continuous cultivation led to a significant reduction of SOC, while the land use transition from grassland to semi-shrubby desert and the progressive succession of natural ecosystems such as semi-shrubby desert and grassland, in contrast, can bring about significant carbon sequestration benefits. The increased water consumption and decreased SOC pool associated with some observed land use changes may induce and aggravate potential ecological risks for both local and downstream ecosystems, including water resource shortages, soil quality declines, and degeneration of natural vegetation. Therefore, it is necessary to balance socioeconomic wellbeing and ecosystem services in land use planning and management for the sustainability of socio-ecological systems across spatiotemporal scales, especially in resource-poor arid environments.

**Keywords** Land use change · Water · Carbon · Arid environment · Ecosystem service · Northwest China

Y. Lü (✉) · Z. Ma · F. Sun · B. Fu
State Key Laboratory of Urban and Regional Ecology, Research Center for Eco-Environmental Sciences, Chinese Academy of Sciences, PO Box 2871, Beijing 100085, China
e-mail: lyh@rcees.ac.cn

Z. Zhao
College of Landscape Architecture and Forestry, Qingdao Agricultural University, 700 Great Wall Road, Chengyang, Qingdao 266109, Shandong, China

## Introduction

Dryland ecosystems, covering approximately 47.2 % of the global land surface, can provide important ecosystem services, including carbon sequestration (Lal 2004). However, dryland ecosystems are vulnerable to human disturbance and are difficult to recover because of low water availability (King et al. 2012). Therefore, trade-offs between carbon and water are particularly remarkable in these ecosystems. Human disturbances are mainly represented as land use changes and land management practices, which have been widely investigated with regard to their soil carbon effects under various geographical contexts and climatic conditions across global drylands (Chen et al. 2010; Romanya and Rovira 2011; Pardo et al. 2012; Santra et al. 2012). The findings have revealed a general trend of soil carbon loss following the conversion of natural ecosystems to agricultural uses without proper management (Haghighi et al. 2010; Romanya and Rovira 2011; Pardo et al. 2012). In this respect, local scale comparative studies examining different land cover types based on soil sampling and analyses have been the mainstream approach in the drylands of China and elsewhere (Li et al. 2006; Su et al. 2010; Raiesi 2012; Saiz et al. 2012; De Baets et al. 2013; Yang et al. 2013). In the current literature, however, quantification of changes in soil carbon and water consumption simultaneously in dryland ecosystems during the process of land use change remains scant, to our knowledge.

China has large areas of dryland ecosystems, within which oases play pivotal roles in supporting local human survival and socioeconomic development. In the oasis-desert ecotones, land use change has mostly been driven by climate variability and socioeconomic demands. The Heihe River Basin (HRB), located in the temperate arid zone of northwestern China, is the second largest inland river basin in China. Significant land use change in the region, resulting in decreases in grassland and water areas and increases in cropland and built-up area, has been identified since the late 1980s (Qi and Luo 2006). Such land use change has pronounced impacts on soil nutrients, water resources, land degradation, and vegetation changes at various scales (Qi et al. 2007; Qi and Luo 2007; Wang et al. 2004, 2007, 2011a). However, most of the relevant studies have focused on the individual impacts of land use change. To the best of our knowledge, no effort has been made to investigate the effects of land use change on water and carbon simultaneously.

The objectives of this paper are: (1) to quantify the main characteristics of land use change in the middle reaches of the Heihe River from 1986 to 2007; (2) to estimate the effects of such land use change on soil organic carbon (SOC) and water consumption through irrigation and activities related to human livelihood; (3) to discuss the implications of the carbon and water effects as well as the pertinent interactions.

## Materials and Methods

### Study Area

The study area is located in the oasis-desert ecotone of the central HRB, in the Zhangye Prefecture of Gansu Province, China. The study area covers 5,890 $km^2$ and had a population of approximately 667,100 in 2011. Its mean annual precipitation is approximately 120 mm, and its mean annual temperature is 6 °C. Geographically, it is located in the central portion of the Hexi Corridor of northwest China (Fig. 1). The main branch of the Heihe River runs through the area and provides a significant share of the area's available water resources. A large portion of this water has been traditionally used by irrigation-based agriculture, which has a history of over 2000 years in the region. As one of the most important food production areas in northwest China, resource consumption in this area can have significant local and basin scale ecological and environmental impacts (Qi and Luo 2007).

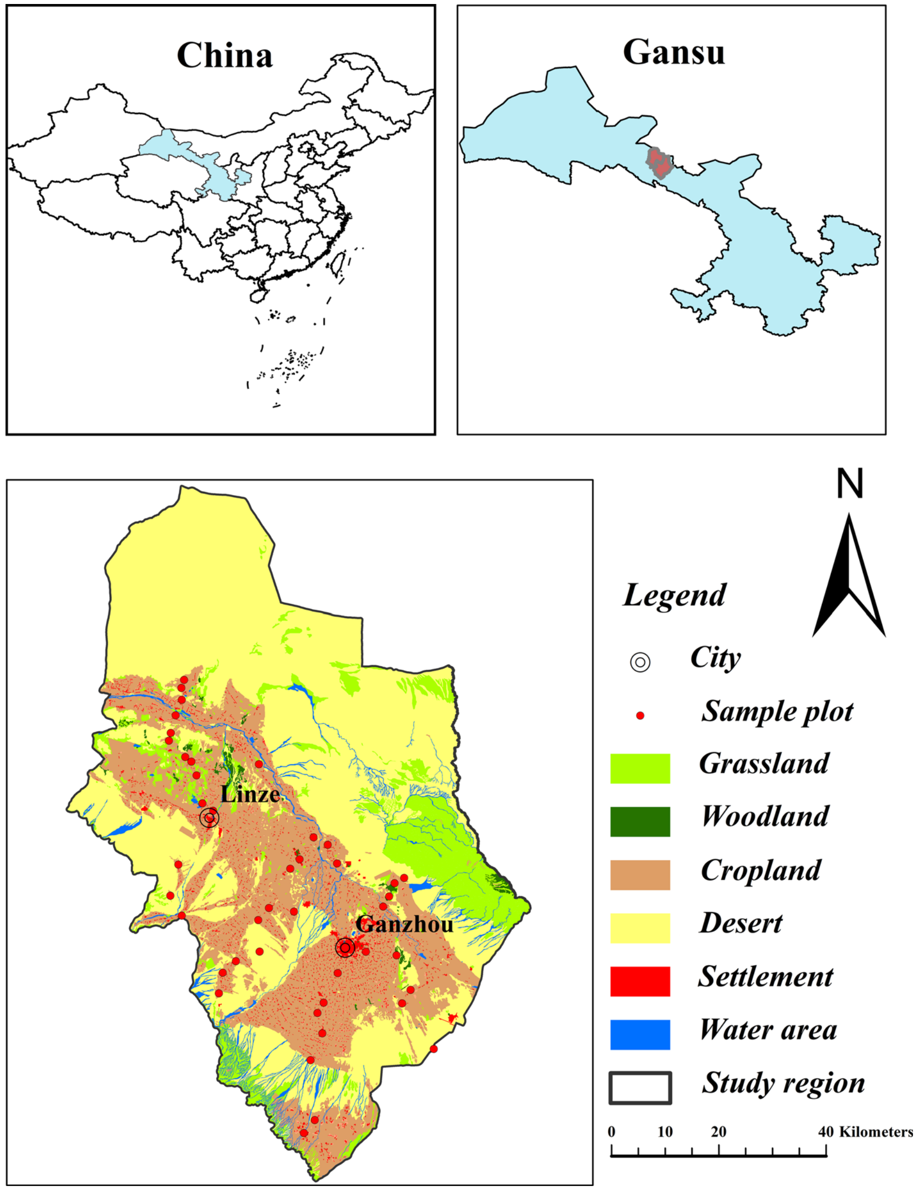
### Land Use Data

Land use data were obtained from the Data Service Center managed by the Cold and Arid Regions Environmental and Engineering Research Institute of the Chinese Academy of Sciences (CAS) (http://westdc.westgis.ac.cn/). Land use maps based on the CAS land cover classification system (Wu et al. 2013) for 4 years (1986, 2000, 2005, and 2007) were collected to form a time series of land use information. Since land use in the HRB has accelerated since 2000 (Li and Yan 2013), in this study we selected land use maps from after 2000 with relative short time intervals. These maps were all derived from Landsat remote sensing images with a ground resolution of 30 m, which are widely used across China for land use change detection and pertinent environmental impact assessment (Shi et al. 2009, 2011). The land use mapping data used in this study were compiled through computer-aided visual interpretation (Liao et al. 2012), and mapping accuracy was evaluated based on sub-pixel fractional error matrices (Latifovic and Olthof 2004). This analysis resulted in an overall landscape mapping accuracy of 95.1 % and a Kappa coefficient of 0.8977 (Liao et al. 2012). Land use characteristics were analyzed quantitatively in the ArcGIS environment. Human settlements are often dominated by impervious land surfaces, under which it is difficult to measure soil carbon levels. Similarly, carbon under bodies of water was not measured. Therefore, human settlements and water bodies were included in the land use change analysis but excluded in terms of their effects on SOC.

### Soil Carbon Storage and Water Consumption

To estimate SOC storage for different land uses, we conducted in situ soil sampling and laboratory analysis. Soil sampling was conducted during the growing season of July for non-cropland land uses and in October for croplands after harvest in 2011 to avoid damaging living crops and the resulting production loss to farmers. A total of 77 soil sampling sites were selected representing the main land uses, including cropland (41 sites), grassland (9 sites), semi-shrubby desert (11 sites), and woodland (16 sites). The number of sampling sites within each type of land use was determined based on the coverage percentage of the land uses in the study area as revealed by the present analysis and similar research (Wang and Liu 2013). At each sampling site, soil samples were collected from six soil layers (0–10, 10–20, 20–40, 40–60, 60–80, and 80–100 cm) at three random locations as replicates using a soil drill in a representative 400-$m^2$ sampling plot. At the same time, a 1-m soil profile was excavated at each site to obtain soil samples for bulk density analysis at the six soil depths using 100-$cm^3$ Kopecki rings along with two other

**Fig. 1** The location and land uses of the study area




sample series obtained by a bulk density drill for surface soil layers up to 40 cm (10 cm for each sample). Subsequently, we obtained 18 soil samples at each site for soil carbon analysis. For the analysis of soil bulk density up to 40 cm in the soil profile, however, we obtained 11 soil samples at each site and put them in 100-cm$^3$ aluminum cases. The soil bulk density was estimated based on the dried weight (oven dried for 24 h at 105 °C) and volume of the soil samples (Choudhary et al. 2013). The soil samples were air dried and sieved through a two-mm mesh for soil carbon analysis. The SOC was determined using the dichromate oxidation method known as the Walkley–Black procedure (Wang et al. 2011b; Fallahzade and Hajabbasi 2012). Beyond the soil data obtained through field sampling and laboratory analyses, the local secondary national soil survey data of the study area were acquired; this survey was completed in 1981 and 1982 (Li et al. 2013; Wang et al. 2013a). In the secondary national soil survey data, the SOC of the top 30-cm soil layer was recorded. Based on these conditions, we computed the average SOC in the three soil layers of 0–10, 10–20, and 20–40 cm as current surface SOC counterparts to estimate the SOC change in comparison to the secondary national soil survey data.

The estimation of SOC storage change was based on the following three hypotheses: (1) the SOC changed dynamically along with time; (2) the dynamic change of the SOC differed significantly among various land use change types; (3) the SOC of top soil (0–30 cm) changes distinctly while the SOC in deep soil layers remains relatively stable because carbon stock change resulting from land use change is a very slow process (Schulp and Verburg 2009; Haghighi et al. 2010; Raiesi 2012). Therefore, we focused on SOC change in the topsoil of typical land use transitions to infer organic carbon effects.

SOC storage change was estimated using the following equation:

$$\Delta \mathrm{SOCD}_{ij} = \Delta \mathrm{SOC}_{ij} \times \mathrm{BD} \times H \quad (1)$$

where BD is the average soil bulk density of the top soil layer (g/cm$^3$), $H$ is the depth of the top soil layer (cm), ΔSOC is the organic carbon content change of the top soil layer (g/kg), and ΔSOCD is the change in SOC in unit volume of soils (kg). Based on this equation and the areas of different land use transitions, the SOC storage change in the top 30-cm soil can be estimated using the following equation:

$$\Delta \mathrm{SOCS}_{ij} = \Delta \mathrm{LU}_{ij} \times \Delta \mathrm{SOCD}_{ij} \quad (2)$$

where $\Delta \mathrm{SOCS}_{ij}$ is the change of SOC storage when land use transformed from $i$ to $j$, $\Delta \mathrm{LU}_{ij}$ is the transition area from land use $i$ to land use $j$, and $\Delta \mathrm{SOCD}_{ij}$ is the change of SOC density along with the land use transition from $i$ to $j$.

Estimation of water consumption focused on cropland and woodland areas that require irrigation and basic human water needs in the study area. The average irrigation water used per ha can be obtained from the local quota system for water use (Li and Zhao 2004; Zhao and Yang 2010). There are some artificial shelter woodlands surrounding the oasis, which also require irrigation for survival but have water requirements much lower than those of croplands (Zhang 2007). The total amount of water consumption can be estimated using the following equations:

$$\mathrm{WU}_{\mathrm{irrigation}} = A_{\mathrm{cropland}} \times Q_{\mathrm{c\_irrigation}} + A_{\mathrm{woodland}} \times Q_{\mathrm{w\_irrigation}} \quad (3)$$

$$\mathrm{WU}_{\mathrm{living}} = P \times Q_{\mathrm{per\_capita}} = P_{\mathrm{rural}} \times Q_{\mathrm{per\_capita\_rural}} + P_{\mathrm{urban}} \times Q_{\mathrm{per\_capita\_urban}} \quad (4)$$

$$\mathrm{WU} = \mathrm{WU}_{\mathrm{irrigation}} + \mathrm{WU}_{\mathrm{living}} \quad (5)$$

WU is water consumption (m$^3$), which contains two aspects: irrigation ($\mathrm{WU}_{\mathrm{irrigation}}$) and livelihood needs ($\mathrm{WU}_{\mathrm{living}}$). $A_{\mathrm{crop}}$ is the area of cropland and $Q_{\mathrm{irrigation}}$ represents water use per hectare for cropland irrigation; $P$ is the human population in the study area divided into rural population ($P_{\mathrm{rural}}$) and urban population ($P_{\mathrm{urban}}$); and $Q_{\mathrm{per\_capita}}$ is the average water use per capita, which also has two parameters $Q_{\mathrm{per\_capita\_rural}}$ and $Q_{\mathrm{per\_capita\_rural}}$. $A_{\mathrm{cropland}}$ and $A_{\mathrm{woodland}}$ were retrieved from land use data, and population data were retrieved from local statistical yearbooks (SBZC 1981, 1989, 2001, 2011).

In order to estimate water consumption through irrigation and basic needs, $Q_{\mathrm{irrigation}}$, $Q_{\mathrm{per\_capita\_rural}}$, and $Q_{\mathrm{per\_capita\_urban}}$ are critical parameters. $Q_{\mathrm{irrigation}}$ for cropland from 2007 to 2012 is 9,660 m$^3$ per hectare (Zhang 2007) and water efficiency of irrigation for the same period is 80 %, while this value is 12,262 and 10,155 m$^3$ in 2000 and 2005, as water efficiency of irrigation in 2000 and 2005 is 50.5 and 55 %, respectively (Luan 2007). $Q_{\mathrm{w\_irrigation}}$ was determined to be 3,375 m$^3$ per hectare according to the local governmental report on ecological water use (ZMG 2003). $Q_{\mathrm{per\_capita\_rural}}$ was assigned at 21.9 m$^3$ (SBZC 2001) as representing a water shortage for rural populations. $Q_{\mathrm{per\_capita\_urban}}$ was set at 36.5 m$^3$ from 2007 to 2012, while $Q_{\mathrm{per\_capita\_urban}}$ was 27.06 and 30.72 m$^3$ in 2000 and 2005, respectively (Luan 2007).

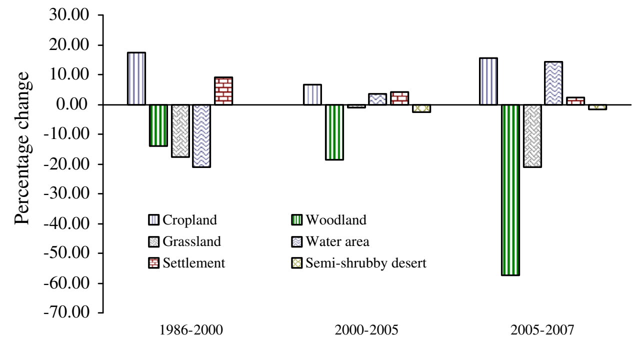



**Fig. 2** Land use change in percentage during various time periods since 1986

## Results

### Land Use Change

The primary land use types examined in the present research are cropland, woodland, semi-shrubby desert land, and grassland. We selected these four land use types as the focus of our analysis, because they account for more than 94 % of the total land surface in the study area. Within these four land use types, croplands and woodlands are highly managed by humans for food production and wind erosion control and thus are most sensitive to changes in population and economic contexts. All four land uses are ultimately dependent on water resource availability influenced largely by climate conditions in arid environment. Results indicated that cropland increased by 45.0 %, whereas woodland, semi-shrubby desert, and grassland decreased by 70.1, 4.1, and 35.7 %, respectively, from 1986 to 2007 (Fig. 2). The other two land use types include human settlement and bodies of water. Human settlement increased by 17.6 % and bodies of water decreased by 7.1 % over the last 22 years. The land use change rate seems to have accelerated from the period of 1986–2000 to 2000–2005 and 2005–2007.

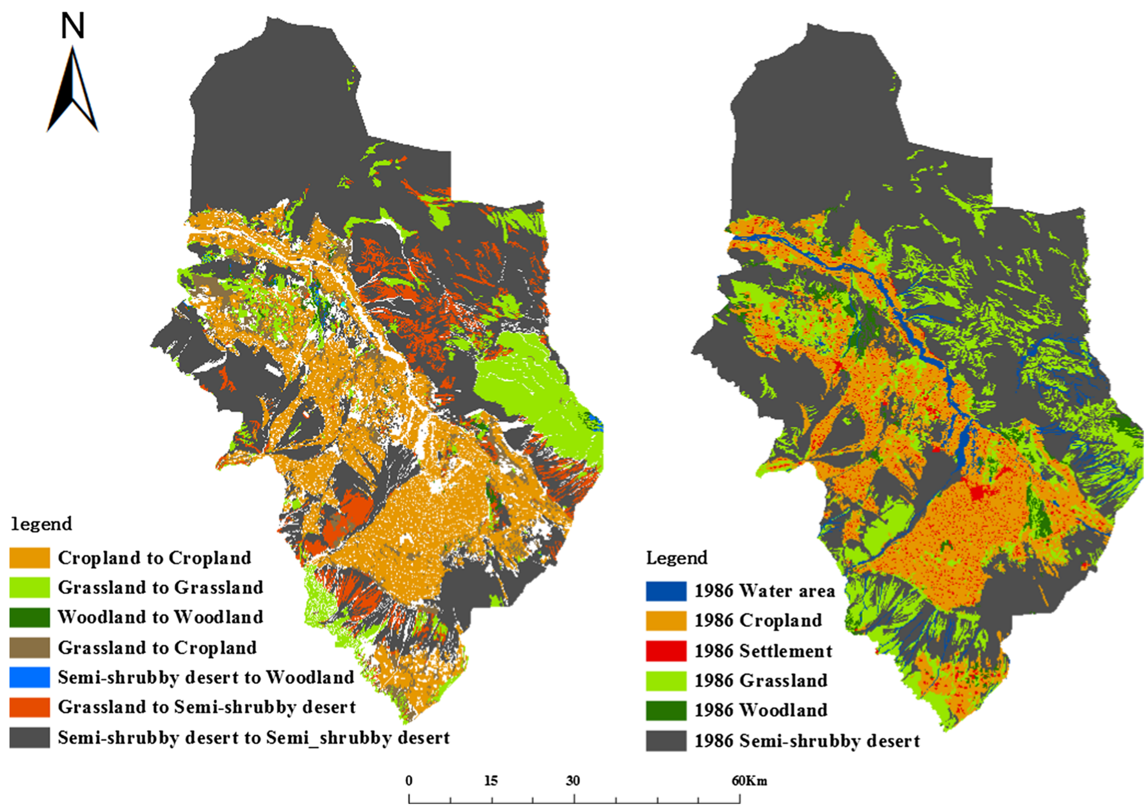



**Fig. 3** The spatial configuration of land use change from 1986 to 2007 (**a**) and the initial land use in 1986 (**b**)

Spatially, the cropland expansion observed from 1986 to 2007 was realized through the encroachment of water area, grassland, woodland, and semi-shrubby desert adjacent to the croplands in 1986 (Fig. 3). The other observed anthropogenic land use conversion is settlement growth at the cost of cropland and the additional three natural or semi-natural land uses (grassland, semi-shrubby desert, and woodland). These types of land use conversions took place in the neighborhood of the former urbanized areas and rural settlements.

## Carbon Distribution Along Soil Profile

For semi-shrubby desert and woodland, the content of organic carbon in the soil profile showed a monotonous decreasing trend with increased soil depth and a slowed decreasing rate slowed under 40 and 60 cm, respectively (Fig. 4). Cropland and grassland shared similar trends in vertical SOC variation of decrease at first and slight increases in the lower soil layers under 80 and 40 cm, respectively.

## Soil Carbon Storage and Change

Results from the laboratory analysis of the soil samples showed that soil under semi-shrubby desert and woodland are among the highest in mean organic carbon content, whereas the lowest average SOC content of all soil profiles was found in grassland and cropland (Table 1). For mean organic carbon storage, woodland and grassland are the highest and the lowest, respectively.

When combining land use change information in the study area, SOC storage in the top soil layer (0–30 cm) decreased by 0.85 g/kg for long-term cropland management, whereas varying degrees of increase were observed for other land use transitions, including grassland converted to cropland, grassland converted to semi-shrubby desert, and maintaining semi-shrubby desert from the early 1980s to 2011–2012 (Table 2). The results also showed that different land use transitions had distinctive SOC impacts. Overall SOC storage in the study area can increase in maintaining large areas of grassland and semi-shrubby desert (Table 2).

## Water Resource Consumption

Table 3 showed that irrigation water used by cropland increased in fluctuation from 1986 to 2007 and water used for basic human needs increased consistently by approximately 1.6 times from 1981 to 2011 (Table 4). It is clear that water used for irrigation accounts for a predominant portion of the total water consumption in this area.

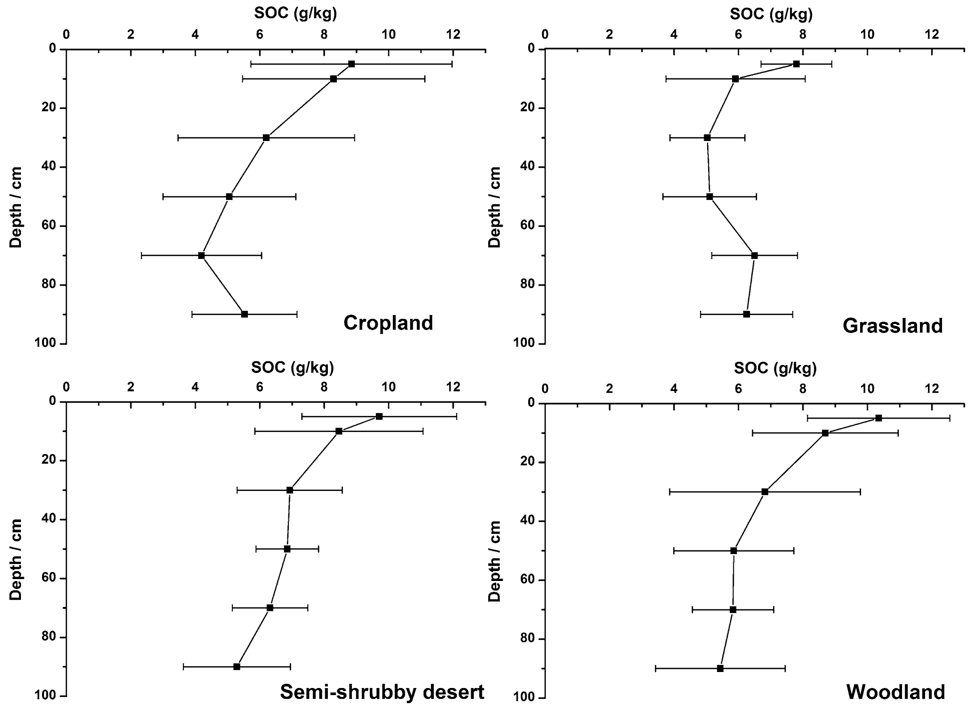



**Fig. 4** The vertical distribution of soil organic carbon in the soil profile. The *horizontal bar* represents the standard error

**Table 1** Mean soil bulk density and soil carbon content and storage under the four focal land uses in 2011–2012 (1-m soil depth)

| Soil characteristics | Cropland ($n = 41$) | | Woodland ($n = 16$) | | Semi-shrubby desert ($n = 11$) | | Grassland ($n = 9$) | |
|---|---|---|---|---|---|---|---|---|
| | Mean | SD | Mean | SD | Mean | SD | Mean | SD |
| Bulk density | 1.47 | 0.10 | 1.49 | 0.16 | 1.22 | 0.18 | 1.41 | 0.12 |
| Organic carbon content (g/kg) | 6.35 | 2.38 | 7.16 | 3.59 | 7.26 | 1.69 | 6.10 | 2.88 |
| Organic carbon storage (kg/m$^3$) | 8.62 | 0.54 | 9.98 | 0.84 | 8.42 | 0.28 | 8.12 | 0.56 |

*SD* standard deviation

Woodlands decreased in the last two decades, resulting in the decrease of related water consumption. Urbanization resulted in a population growth trend from 1981 to 2011 in towns and cities, and urban residents consume more water during the course of daily life than rural inhabitants do. In summary, irrigation water consumption increased by 4.15 % from 1986 to 2007. In 2007, the total irrigation water consumed by cropland and woodland was over 1.7 billion cubic meters, accounting for approximately 82.4 % of the average total water coming from the upper reaches of the Heihe River (Bi et al. 2013).

## Discussion

### Potential Ecological Risks Concerning SOC Change

Land use has been identified as one of the most powerful agents driving environmental change. It also profoundly impacts ecosystems in terms of their capacity to provide goods and services for the survival and development of human society. The most prominent land use change in the study area during the past two decades has been the agricultural expansion coming at the expense of natural and semi-natural land use including in grassland, woodland, and semi-shrubby desert. Numerous studies have focused on SOC due to its importance in global change (Lü et al. 2012). These land use transitions have negative impacts on carbon sequestration by dryland ecosystems in the study area in terms of SOC (Table 2), which may induce soil quality deterioration and higher risks of wind erosion and desertification at local scales. These negative effects may be the result of decreased retention of organic carbon (Cusack et al. 2013) and acceleration of SOC decomposition (Arroita et al. 2013) in agricultural fields compared to natural ecosystems in dryland areas, where SOC is a determining factor in soil quality and land productivity (Srinivasarao et al. 2013). Lack of ground cover protection

**Table 2** Soil organic carbon storage change among typical land use change modes

| Change mode | Variance test | *n* | Average change (g/kg) | SE | Bulk density (g/cm$^3$) | Area of transform (km$^2$) | Total change by category (Gg) |
|---|---|---|---|---|---|---|---|
| Woodland to woodland | a | 9 | −6.36 | 1.72 | 1.31 ± 0.12 | 33.32 | −83.28 |
| Semi-shrubby desert to cropland | ab | 5 | −5.35 | 1.38 | 1.39 ± 0.08 | 165.45 | −369.11 |
| Semi-shrubby desert to woodland | ab | 7 | −2.68 | 1.84 | 1.40 ± 0.07 | 1.35 | −1.52 |
| Cropland to cropland | ab | 31 | −0.85 | 0.90 | 1.40 ± 0.09 | 1,222.10 | −436.29 |
| Grassland to cropland | ab | 5 | 0.52 | 1.27 | 1.47 ± 0.12 | 297.54 | 68.23 |
| Grassland to grassland | b | 8 | 1.37 | 1.14 | 1.45 ± 0.13 | 619.86 | 568.94 |
| Grassland to semi-shrubby desert | b | 5 | 1.71 | 1.28 | 1.16 ± 0.05 | 81.14 | 48.28 |
| Semi-shrubby desert to Semi-shrubby desert | b | 6 | 3.03 | 0.88 | 1.27 ± 0.18 | 2,591.00 | 2,991.12 |

Duncan's multiple range test, lowercase letter represents at 0.05 test level; *SE* standard error; homogeneity test sig. = 0.712, variance test sig. = 0.025

**Table 3** Water consumption by irrigation

| Year | Cropland area (km$^2$) | Average water use by cropland (m$^3$/ha) | Water use by cropland (Gt) | Area of woodland (km$^2$) | Average water use by woodland (m$^3$/ha) | Total water use (Gt) |
|---|---|---|---|---|---|---|
| 1986 | 1,222.10 | 12,705 | 1.553 | 116.13 | 3,375 | 1.592 |
| 2000 | 1,435.02 | 12,262 | 1.760 | 99.85 | | 1.793 |
| 2005 | 1,531.38 | 10,155 | 1.555 | 81.37 | | 1.583 |
| 2007 | 1,771.27 | 9,660 | 1.711 | 34.67 | | 1.723 |

*Source* average water use by cropland per hectare is from Luan (2007) and Zhang (2007); average water use by woodland data is from ZMG (2003)

**Table 4** Changes in total water consumption for basic human needs

| Living water | Rural population/ten thousand | Urban population/ten thousand | Water use ($10^7$ t) |
|---|---|---|---|
| 1981 | 41.84 | 7.485 | 1.12 |
| 1989 | 44.16 | 10.52 | 1.25 |
| 2001 | 47.88 | 15.44 | 1.47 |
| 2011 | 45.26 | 21.45 | 1.77 |

Quotas for human water consumption used here were developed based on local government statistical information (SBZC 1981, 1989, 2001, 2011)

**Table 5** Soil organic carbon levels of newly reclaimed and long-term croplands

| Soil layers | New: g/kg ($n = 10$) | Long-term: g/kg ($n = 31$) |
|---|---|---|
| 0–10 | 6.03 ± 2.50 | 9.53 ± 2.80 |
| 10–20 | 5.62 ± 2.95 | 8.92 ± 2.26 |
| 20–40 | 3.26 ± 1.57 | 6.88 ± 2.21 |
| 40–60 | 2.81 ± 1.37 | 5.57 ± 1.90 |
| 60–80 | 2.18 ± 1.45 | 4.63 ± 1.38 |
| 80–100 | 4.86 ± 1.26 | 5.75 ± 1.92 |

in windy seasons and decreasing organic carbon in soils are the main sources of risks with regard to wind erosion and desertification for croplands in the study area. The decrease of SOC may contribute to the deterioration of soil structural integrity and various functions that can lead to erosion and desertification problems, as revealed by studies conducted in North American and Mediterranean dryland areas (Li et al. 2008; Maestre and Puche 2009; Garcia-Orenes et al. 2012). Furthermore, decreased SOC storage capacity brought about by cropland expansion at local scales in dryland ecosystems weakens the high potential for these areas as carbon sinks, with far reaching implications for global change (Lal 2004).

For some unprivileged land cover types with poor soil fertility, such as found in sandy deserts or the Gobi desert, cropland cultivation does have some positive effects in improving SOC, as observed in the present study (Table 5) and in similar studies (Fallahzade and Hajabbasi 2012; Yang et al. 2013). However, these effects can only be considered significant on a long time scale with intensive management and resources input, including fertilization and irrigation (Su et al. 2010). Compared to the carbon

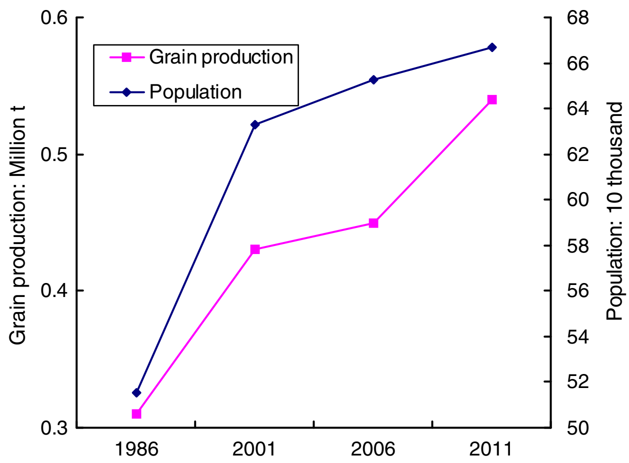



**Fig. 5** Grain production in the study area since 1986

sequestration effects of natural ecosystems during progressive succession, such as the grassland and semi-shrubby desert ecosystems in this research (Table 2), the stability, long-term cost effectiveness, and sustainability of positive SOC effects from croplands remain questionable (Su et al. 2010; Yang et al. 2013).

## Land Use Management Toward Sustainability in Drylands

So far, it is clear that SOC storage and water budgets have been disturbed significantly, driven by increased agricultural production and economic return and represented by the large scale expansion of croplands. This research suggests that the relationships among ecosystem services such as grain production, water consumption, and carbon sequestration can be significantly modified by land use change at the regional scale.

In the middle reach of the Heihe River, the consistent expansion of cropland may be driven by the demand for food and economic return resulting from the steady growth of the local population, which increased by 5.5 % from 1996 to 2006 (Bing et al. 2010); this figure was 19.4 and 22 %, respectively, for the periods of 1989–2006 and 1989–2011 as calculated from local governmental statistics (Fig. 5) (SBZC 1989, 2011). Feng (2010) investigated long-term (1949–2005) grain production in the study area and found that cropland acreage and irrigation coverage are primary factors responsible for the fluctuation of total grain production. Therefore, cropland is strongly related to grain production and economic return. For example, the net economic return from maize cropland in 2000 was 12,000 RMB or 1,446 US Dollars per hectare (Zhao et al. 2002), which is a substantial economic return for local farmers. As revealed by local governmental statistics, grain production in the study area increased by 43.7 and 72.6 % from 1986–2006 and from 1986–2011 (Fig. 5), respectively (SBZC 2001, 2011). From the present research, it is clear that this gain in grain production and economic return resulted in the losses of large areas of natural and semi-natural ecosystems (Fig. 2). With the changes in land uses, at least 0.13 billion cubic meters of more water was used for irrigation in 2007 than in 1986.

For irrigation water use estimation, the parameters were set at the optimal intensity of 500–600 mm for maize dominant crops (Li et al. 2012) and 300–400 mm for woodland, which has been adopted for water resource assessment and management in the study area. The woodlands are generally poplar-based shelters used as windbreaks that require irrigation (Su et al. 2010; Zhao et al. 2010). However, the actual water used by farmers for irrigation was much greater than the optimal level because of a flood irrigation approach (Ji et al. 2007). The reported household survey data demonstrated that cropland irrigation water use can reach 21 thousand $m^3$ per hectare (Zhang et al. 2014) with low water use efficiency and that over 75 % of irrigation water was wasted, accordingly (Liu and Li 2012). Therefore, this study underestimated irrigation water consumption but supported the existence of high water saving potential in the study area. It is also clear that practical approaches to reducing water consumption by irrigation include increasing water use efficiency and cutting off the unit area water use (Table 3) through the improvement of market, management, and technical measures (Wang et al. 2012; Ge et al. 2013; Zhang et al. 2013; Zhang et al. 2014).

Water is one of the most important natural resources for ecosystem health and human survival in arid regions. The increased water consumption resulting from irrigation will potentially limit the water supply to other sectors as well as the amount of available water for downstream areas. Such changes in water resource allocation will have significant impacts on the long-term sustainability of the socio-ecological systems in the middle and lower reaches of the Heihe River, as water-related environmental degradation problems have been observed in this region (Qi and Luo 2007), including natural vegetation dieback of up to 80 % and surface runoff decreases of up to approximately 30 % (Wang et al. 2007), groundwater depletion (Wei et al. 2007), desertification, secondary salinization incurred by over-irrigation (Ma et al. 2013), and agricultural non-point source water pollution (Nan et al. 2010), which were also widely observed in other dryland areas such as those existing in Australia and Argentina (Beverly et al. 2011; Jayawickreme et al. 2011; Kienzler et al. 2012).

Conversions of natural ecosystems by anthropogenic activities have potentially undermined the capacities of ecosystems to provide supporting and regulating services that are usually beyond economic scale and crucial for environmental sustainability (DeFries et al. 2004; van Oudenhoven et al. 2012). Our research indicated that in water resource-limited ecosystems, although change of land use has the potential to create higher economic return, it may simultaneously incur ecological risks on watershed health and carbon cycling in both local and nearby systems. Once these ecosystems have been negatively impacted, a sustainable land management plan made with full consideration of human–environment interactions and socio-economic drivers should be adopted to help the recovery of the disturbed ecosystem (Cowie et al. 2011; Eppink et al. 2012; Nkonya et al. 2011; Schwilch et al. 2011), despite the fact that some of the ecosystem services may be permanently lost (Benayas et al. 2009). Therefore, management of integrated watersheds should put water resources as the central consideration, as water is a basic resource that constrains many other socio-ecological systems. Different land use arrangements and land management practices can make profound differences in terms of water consumption (Bowmer 2011; Darradi et al. 2012).

Subsequently, it is necessary to improve land use planning and management for watershed health and sustainability in the HRB. Firstly, extensive cropland expansion encroaching natural ecosystems needs to be curbed for environmental protection and water resource conservation. In this respect, local governmental agencies in charge of ecosystem and land management should attach more importance to ecosystem monitoring and land resource surveys meant to find and correct non-sustainable land resource development practices including conversion of natural ecosystems to croplands or other human dominant land use types. Secondly, marginal croplands with low production but high water resources and nutrient input are better for setting aside for ecological restoration supported by suitable payment for environmental service schemes (Wang et al. 2013b). This can bring direct benefits with regard to water resource conservation, carbon sequestration, and soil quality improvement. Thirdly, organic and conservation farming practices need to be promoted on high-productivity croplands with optimized crop planting patterns (Nan et al. 2010; Zeng et al. 2012) that cut down unit area water use and improve ecological soil functions without undermining economic profits. Finally, integrated approaches need to be adopted, which may include the improvement of infrastructure and technology in agriculture, enhanced and effective ecological conservation and restoration, and accounting for the status and trends of ecosystem services as well as their relations with human welfare.

## Conclusions

Socioeconomic development has resulted in significant land use change represented by significant cropland and human settlement expansion along with the shrinkage of woodland, grassland, and semi-shrubby desert in the middle reach of the Heihe River from 1986 to 2007. Although this type of land use change has brought with it higher grain production and economic return for the local people, it has also resulted in increased water consumption and significant risk of SOC loss. The increased consumption of fresh water from the middle reach of the Heihe River may bring the risk of environmental degradation on site and at the downstream regions. Therefore, our research calls for an action to simultaneously monitor land use and ecosystem service change for effective land management decision-making. In arid environments where supporting resources such as water are usually limited, land use decisions may require incorporating trade-offs among land use objectives, interactions among ecosystem services, and complex human–environment interactions to make the land management more adaptive across various spatiotemporal scales.

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