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# Impacts of LUCC on soil properties in the riparian zones of desert oasis with remote sensing data: A case study of the middle Heihe River basin, China

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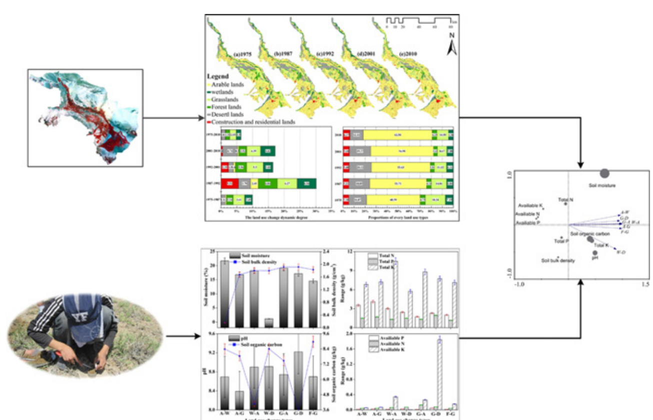
## HIGHLIGHTS

- We analyzed the impacts of LUCC on soil properties in the riparian area zones of desert oasis.
- The combination of soil experiment with RS images in a long time scale
- Soil moisture and soil organic carbon can be explained by LUCC well.
- Soil nutrients have no significant correlation with LUCC.

## GRAPHICAL ABSTRACT



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## ABSTRACT

Large-scale changes in land use and land cover over long timescales can induce significant variations in soil physicochemical properties, particularly in the riparian zones of arid regions. Frequent reclamation of wetlands and grasslands and intensive agricultural activity have induced significant changes in both land use/cover and soil physicochemical properties in the riparian zones of the middle Heihe River basin of China. The present study aims to explore whether land use/land cover change (LUCC) can well explain the variations in soil properties in the riparian zones of the middle Heihe River basin. To achieve this, we mapped LUCC and quantified the type of land use change using remote sensing images, topographic maps, and GIS analysis techniques. Forty-two sites were selected for soil and vegetation sampling. Then, physical and chemical experiments were employed to determine soil moisture, soil bulk density, soil pH, soil organic carbon, total nitrogen, total potassium, total phosphorous, available nitrogen, available potassium, and available phosphorous. The Independent-Samples Kruskal–Wallis Test, principal component analysis, and a scatter matrix were used to analyze the effects of LUCC on soil properties. The results indicate that the majority of the parameters investigated were affected significantly by LUCC. In particular, soil moisture and soil organic carbon can be explained well by land cover change

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and land use change, respectively. Furthermore, changes in soil moisture could be attributed primarily to land cover changes. Changes in soil organic carbon were correlated closely with the following land use change types: wetlands–arable, forest–grasslands, and grasslands–desert. Other parameters, including pH and total K, were also found to exhibit significant correlations with LUCC. However, changes in soil nutrients were shown to be induced most probably by human agricultural activity (i.e. fertilize, irrigation, tillage, etc.), rather than by simple conversions from one land use/cover types to the others.



## 1. Introduction

Land use and land cover change (LUCC) is a basic parameter used to quantify changes in the natural environment and human activity (Mendoza et al., 2011; Turner et al., 2007). Generally speaking, LUCC is considered as the different types of modification that human did on the Earth's surface (Arsanjani, 2012; Ellis, 2013). In which, land cover change is defined as the conversions between different physical and biological covers (Ellis, 2013; Lambin et al., 2001). However, land use change mostly refers to the alternations of land surface processes (Ellis, 2013; Lambin and Meyfroidt, 2011). As a crucial component of global change and sustainable development, LUCC has major effects on ecosystems, climate change, and grain yield, among other things (Abd El-Kawy et al., 2011; Foley et al., 2005; Jansen et al., 2008; Leifeld, 2013). Moreover, LUCC clearly reflects environmental crises and regional ecological safety (Alemayehu et al., 2009; Mendoza et al., 2011; Zhang et al., 2010). In particular, LUCC is known to affect soil nutrients and result in soil erosion and even soil degradation (Ali, 2006; Covaleda et al., 2011; Fu et al., 2001; Gamboa and Galicia, 2011; Liu et al., 2010b; Maquere et al., 2008; Shirvani et al., 2010).

As the transition zones between terrestrial and aquatic systems, riparian zones play a key role in maintaining river ecosystem health (Tang et al., 2014). In particular, riparian zones perform many ecological functions, including preserving biodiversity and purifying polluted waters (Batlle-Aguilar et al., 2012). However, they are also fragile systems and can be modified easily by land use change (Hlubikova et al., 2014; Krause et al., 2008). Over the past century, the scale and intensity of land use and exploitation have increased dramatically, particularly in arid regions. This increased exploitation eventually led to increased soil erosion, soil salinization, and soil nutrient loss, all of which have induced dramatic reductions in soil quality (Abbasi et al., 2007; Caravaca et al., 2002; Majaliwa et al., 2010). However, it is well known that soil properties such as soil moisture and nutrient content are key components affecting ecosystems and agricultural development in arid environments (Majaliwa et al., 2010; Yang et al., 2012); therefore, such properties play a key role in sustainable development in arid areas. In this context, investigation of the effects of LUCC on soil properties in arid areas is essential. Consequently, many previous studies have investigated the impacts that LUCC can have on soil properties and soil erosion in arid regions (Chen and Peng, 2000; Moges et al., 2013; Quan et al., 2011; Wang et al., 2004; Yang et al., 2008; Zhang et al., 2013). For example, Quan et al. (2011) and Yang et al. (2008) found LUCC to be the primary cause of soil erosion in the Liupan Mountains and the upper reaches of the Shiyang River. Similarly, Moges et al. (2013) found that different types of land use change led to variable declines in soil quality in southern Ethiopia. Moreover, Chen and Peng (2000) asserted that, while rational land use could prevent land degradation, irrational land use leads to the decline of soil productivity in arid regions.

In desert oasis areas, riparian zones are typical sites of concentrated human activity owing to their fertile soil and rich irrigation water. As a result, soil properties are vulnerable to the impacts of the LUCC induced by various human activities. The Heihe River is one of the most important inland rivers in the northwest arid district of China. Owing to rapid economic and population growth in the region, the river's ecological environment has deteriorated rapidly in recent years, particularly in the riparian zones of the middle reaches. These riparian zones represent the core area of socio-economic development in the Heihe River basin; accordingly, land use here is dominated by high-density human activity. With increasing economic development and population inflation, ecological problems such as depression of the groundwater level, destruction of forest, acceleration of desertification, and degradation of wetlands have become increasingly serious (Kang et al., 2007; Li and Zhao, 2010; Qin et al., 2011).

Therefore, understanding the effects of LUCC on soil properties in the middle reaches of the Heihe River is of considerable importance for the sustainability of the region. To help achieve this, we used remote sensing data for 1975–2010 and experimental soil physicochemical data obtained in 2011 to investigate how LUCC affects soil properties in this arid inland basin. Here, we discuss the effects of land use and land cover change on soil properties. In this way, we hope that our study can provide help for constructing a right mode of land resources utilization and agriculture development in the semi-arid and arid areas. Moreover, the analysis of the impacts that LUCC did on soil properties also can be seen as a particular reference to the possible soil degradation.

## 2. Study area

The middle Heihe River basin is located in the western part of Gansu Province, northwestern China (Fig. 1). It has been a rich and livable place since ancient times. According to official statistics, the total population in the middle Heihe River basin was soaring from $58.67 \times 10^4$ in 1975 to $82.55 \times 10^4$ in 2010. The excellent natural condition is the key to this region to continue its prosperity. The terrain here can be divided into mountains and plains, with dominant topographic features including the Hexi Corridor plain and the Qilian and Heli Mountains (located to the south and north of the Heihe River, respectively). The altitude spans 1234–3633 m above sea level, with maximum altitude in the Qilian Mountains. From the mountain to the plain, the sediments tend to be finer. The lithology varies from coarse-grained gravel pebbles with a thickness of 1000 m to fine-grained sand and silt with a thickness of 50–200 m gradually (Fan, 1991; He et al., 2013). Flat terrain, fertile soil and sufficient labor force, have made this region to be the great place for agricultural activities, especially in the riparian zone.

By coupling the relief degree of the land surface (RDLS) and soil types (only flood alluvial soils were taken into consideration) with elevation, we extracted the boundary of the riparian zone in the middle Heihe River basin; this zone was found to cover an area of 2378.46 km$^2$, which account for 22.12% of the total area of the middle Heihe River basin (Fig. 1). The climate of this region is characterized by cold winters, hot summers, and generally dry weather. Mean annual precipitation and mean annual evaporation in the region are 50–250 mm and 2000–3500 mm, respectively. Crops and forests are distributed throughout the piedmont alluvial fan and floodplain, with arable land and grassland, and the Gobi desert constituting the dominant land use types (with less abundant temperate shrubs and desert vegetation). Soils in the study area are typically Aridisols and Entisols (Su, 2007). However, the soils along the Heihe River have often been developed into Siltigi–Orthic Anthrosols owing to long-term agricultural activity (Su and Yang, 2008). Nevertheless, because the sampling sites for the present study are distributed along the Heihe River, differences in soil type between samples should be minimal.

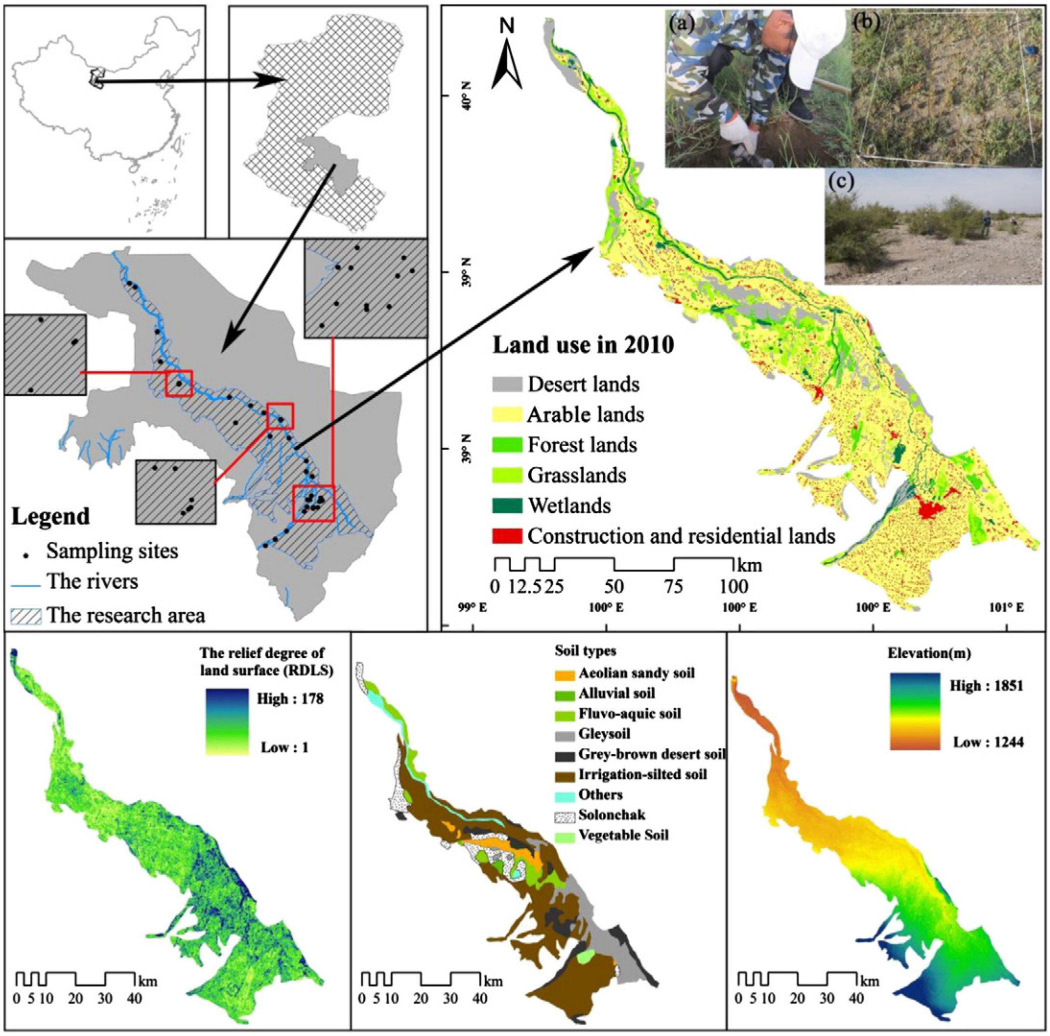


**Fig. 1.** The research area (a: soil sampling, b–c: vegetation sampling).

## 3. Materials and methods

The present study utilized materials including remote sensing images and experimental data. The specific workflows for the data acquisition and processing steps are illustrated in Fig. 2.

### 3.1. Data sources

LUCC information in the middle reaches of the Heihe River during 1975–2010 was extracted from remote sensing images (Table 1). Image years were selected based on historical events (i.e., the initiation of reform/openness and western development in the 1980s and 1990s, respectively), policy changes (i.e., the initiation of water diversion work in the middle Heihe River basin in 2001), and image availability. Image months were selected based primarily on the flourishing time of vegetation (i.e., summer and the start of autumn). Interpreted images were combined with LUCC data at a scale of 1:100,000 (for 1985 and 1996), which were acquired from the Chinese Academy of Sciences (http://westdc.westgis.ac.cn/). A topographic map at a scale of 1:50,000 (produced by field classification in 1972) was also combined with the interpreted images.

### 3.2. Processing of remote sensing images

Image processing was conducted using ERDAS Imagine (version 9.2). The data preprocessing steps adopted in the present study included adjustment, joining, enhancement, and cutting of images. Additionally, the MSS (58 m), TM (30 m), and ETM+ (30 m) images were resampled to a $60 \times 60$ m pixel size using the nearest-neighbor resampling technique. Artificial visual interpretation was applied to the interpretation of land use information based on sufficient field survey and reference data. Then, five land use/land cover types were identified according to the classification criteria of Liu et al. (2010a).

The interpretation accuracy of land use information from 1975, 1987, and 1992 depends on the accuracies of the topographic map produced in 1972 and the land use data for 1985 and 1996, respectively. Since all of these are known to possess high accuracy, we focused only on assessment of the accuracy of the classifications for 2001 and 2010; we assessed the accuracy of interpretation of these images based on field survey and high-resolution images. The kappa coefficient $K$ was calculated as follows:

$$K = \frac{N \sum_{i=1}^{m} P_{li} - \sum_{i=1}^{m} \left( P_{pi} \times P_{li} \right)}{N^2 - \sum_{i=1}^{m} \left( P_{pi} \times P_{li} \right)}, \quad (1)$$

where $N$ is the total number of samples and $P_{pi}$ and $P_{li}$ are the total numbers of samples in the row and rank, respectively, of a given land use and land cover type. The results demonstrate that the accuracies of classified images for 2001 and 2010 were 84.06% and 81.18%,

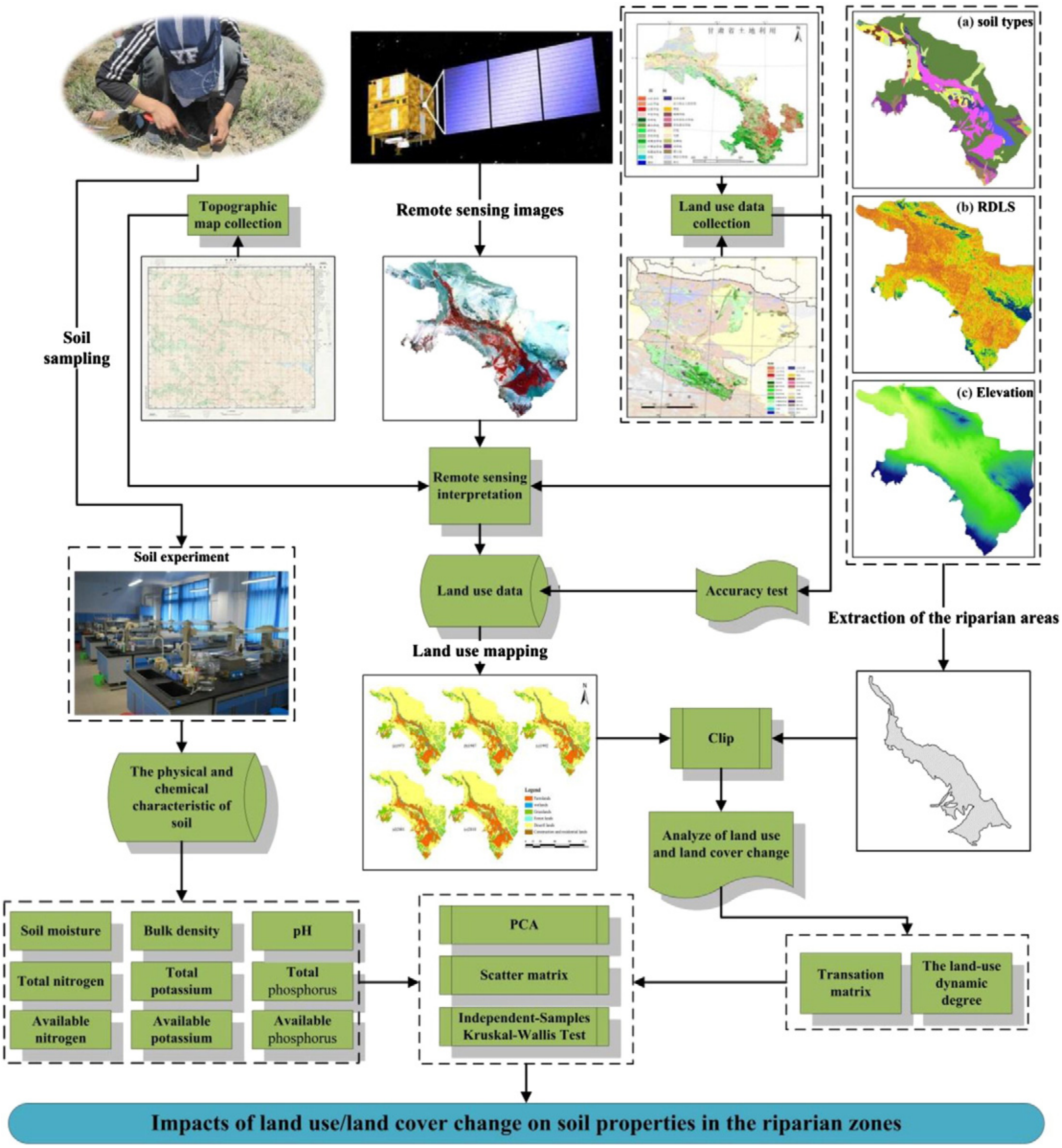


**Fig. 2.** Flowchart illustrating the methodology adopted in the present study.

respectively (Table 2). Based on the processed remote sensing data, five time series of land use data were acquired for analysis in the present study (Fig. 3).

### 3.3. Analysis of land use change

The transition matrix and land use dynamic degree proposed by Liu et al. (2014) were selected to analyze the process of the land use change. These methods can indicate dynamic changes of one land use type over a certain period. Moreover, they can well illustrate processes of spatiotemporal change in land use. The land use dynamic degree of each land use type was calculated according to the following equation:

$$S=\left\{\sum_{ij}^{n} S_{ij}/S_i\right\}\times\left(\frac{1}{t}\right)\times 100\%, \quad (2)$$

where $S_i$ is the area of land use type $i$ in the starting year, $\Delta S_{i-j}$ is the area of land use type $i$ converted into other types, $t$ is the change period, and $S$ is the land use dynamic degree in the period $t$.

**Table 1**
Data sources for landscape information for study area.

| Year | Data source type | Remote sensing image |
|---|---|---|
| 1975 | MSS | 143/33, 1975.10.07; 145/32, 1975.10.09; 144/33, 1976.07.04 |
| 1987 | TM | 133/33, 1987.08.15; 134/33, 1987.10.09; 134/32, 1989.09.28 |
| 1992 | TM | 133/33, 1992.09.05; 134/33, 1992.08.27; 134/32, 1991.09.02 |
| 2001 | ETM | 133/33, 1999.07.07; 134/33, 2001.07.03; 134/32, 2001.08.20 |
| 2010 | ETM | 134/33, 2010.08.05; 133/33, 2010.08.14; 134/32, 2010.08.21 |

**Table 2**
The error matrix for the process of interpreting 2001 and 2010 data.

| GPS and Google Earth samples \ Classification | $A_1$ | $A_2$ | $A_3$ | $A_4$ | $A_5$ | $A_6$ | $A_7$ | $A_8$ | $A_9$ | $A_{10}$ | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|
| $A_1$ | 21/1 | 2/1 | 2/0 | 0 | 0/5 | 0 | 0 | 0 | 0 | 0 | 25/7 |
| $A_2$ | 1/0 | 61/8 | 2/0 | 6/0 | 0 | 0 | 0 | 0 | 0 | 0 | 70/8 |
| $A_3$ | 2/0 | 7/1 | 18/2 | 1/0 | 0 | 0 | 0 | 0 | 0 | 0 | 28/3 |
| $A_4$ | 1/0 | 1/0 | 1/0 | 39/5 | 0 | 0 | 0 | 0 | 0 | 0 | 42/5 |
| $A_5$ | 0 | 0 | 0 | 0 | 16/191 | 0 | 0 | 0 | 0 | 0 | 16/191 |
| $A_6$ | 0 | 0 | 0 | 0 | 0 | 3/0 | 0 | 0 | 0 | 0 | 3/0 |
| $A_7$ | 0 | 2/0 | 0 | 3/0 | 0 | 0 | 13/2 | 0 | 0 | 0 | 18/2 |
| $A_8$ | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3/0 | 1/0 | 0 | 4/0 |
| $A_9$ | 1/0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2/0 | 0 | 3/0 |
| $A_{10}$ | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 8/1 | 8/1 |
| Total | 26/1 | 73/10 | 23/2 | 49/5 | 16/196 | 3/0 | 13/2 | 3/0 | 3/0 | 8/1 | 217 |

Notes: (1) $A_1$: arable land, $A_2$: grassland, $A_3$: forest, $A_4$: desert, $A_5$: construction, $A_6$: river, $A_7$: bottomland, $A_8$: lake, $A_9$: marsh, $A_{10}$: reservoir and pond. (2) The left column and top row include sample numbers for 2010 and 2001, respectively.

### 3.4. Field survey and experimental design

For our research, we adopted the concept of "space for time" proposed by Li et al. (2007) to compensate for the lack of historical data. Our field survey was conducted in summer (July–August) 2011 with the help of local guides and LUCC data acquired from remote sensing images. During this period, we avoid the raining time and the busy farming season. In this way, we did our best to reduce the interference effects of human being activities (i.e., plow, fertilization, irrigation, etc.) and natural factors on soil properties. Moreover, the land cover information of the study area in this season is the best period and it can help us to acquire a more accurate land cover information. In total, 42 randomly selected sampling sites (Table 3) were aligned primarily parallel to the river. GPS was used to fix the position of each sampling site, and basic site characteristics such as land use type, height, and distance to the river were recorded. Of the selected sites, 16 had experienced changes in land use and 26 had not (according to the interpreted satellite data). After sampling sites were determined, 2–4 quadrats (each 1 × 1 m) were set for herbaceous plants and 1–2 quadrats of 30 × 30 m were set for trees or shrubs.

Vegetation characteristics (including type, number of plants, plant height, coverage, and density) were recorded before soil sampling to allow investigation of changes in plant communities. Then, soil samples for physicochemical analysis were obtained using a soil auger and cutting ring. These samples were collected along the diagonal lines of quadrats at three depths (0–10, 10–20, and 20–40 cm).

### 3.5. Experimental analysis of soil and vegetation classification

First, conventional methods were used to determine the physicochemical properties of soil samples (e.g., soil moisture, soil bulk density). The potassium dichromate titration method, the micro-Kjeldahl method, the sulfuric acid–perchloric acid heating digestion method, and the alkali fusion–flame photometry method were used to determine organic matter content, total nitrogen, total phosphorous, and total potassium of soil, respectively. A flow analyzer (SKALAR 8505) was used to determine the available nutrient content in soil. Soil bulk density and moisture were determined using the cutting ring method.

Vegetation was categorized based on the constructive species of the community, with the woody, tall herbaceous (i.e., higher than 60 cm),

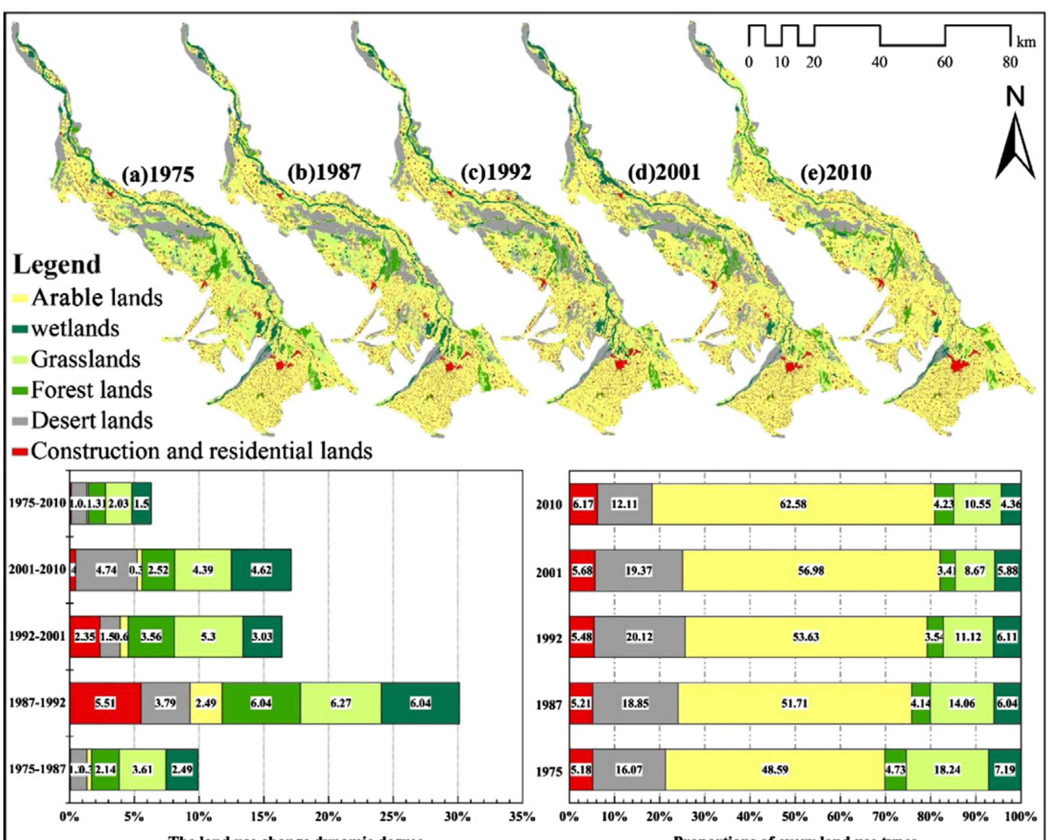


**Fig. 3.** Land use and land cover change in the study area during 1975–2010.

**Table 3**
Descriptions of the sampling sites.

| Site | Lat | Lon | Elevation (units: m) | Dominant vegetal species | Distance to the nearest river (units: m) | Distance to the nearest settlement (units: m) | Distance to the nearest road (units: m) |
|---|---|---|---|---|---|---|---|
| $A_1$ | 38.97°N | 100.44°E | 1417.20 | *Agropyron cristatum (Linn.) Gaertn.* | 2187.95 | 818.60 | 101.97 |
| $A_2$ | 38.88°N | 100.31°E | 1533.00 | *Taraxacum mongolicum Hand.-Ma;zz.* + *Elymus dahuricus Turcz.* | 243.44 | 549.82 | 2456.87 |
| $A_3$ | 38.84°N | 100.26°E | 1576.10 | *Typha orientalis Presl.* | 10.43 | 332.76 | 22.84 |
| $A_4$ | 38.84°N | 100.26°E | 1578.40 | *Agropyron cristatum (Linn.) Gaertn.* | 38.46 | 286.52 | 29.44 |
| $A_5$ | 38.82°N | 100.21°E | 1639.90 | *Chenopodium glaucum* | 620.24 | 773.39 | 34.51 |
| $A_6$ | 38.95°N | 100.40°E | 1442.30 | *Tamarix arceuthoides* + *ArtemisiaordosicaKrasch.* | 213.49 | 877.71 | 16.94 |
| $A_7$ | 38.97°N | 100.41°E | 1425.10 | *Elaeagnus angustifolia Linn.* + *Agropyron cristatum (Linn.) Gaertn.* | 415.12 | 1446.03 | 171.64 |
| $A_8$ | 38.99°N | 100.42°E | 1403.30 | *Phragmites australis (Cav.) Trin. ex Steud* | 265.5 | 1202.33 | 131.4 |
| $A_9$ | 39.01°N | 100.43°E | 1448.00 | *Stipa capillata Linn.* | 46.08 | 847.23 | 1516.5 |
| $A_{10}$ | 39.08°N | 100.43°E | 1436.00 | *Elaeagnus angustifolia Linn.* | 1202.00 | 84.20 | 18.89 |
| $A_{11}$ | 39.10°N | 100.40°E | 1370.80 | *Polygonum lapathifolium L.* | 27.50 | 423.98 | 260.75 |
| $A_{12}$ | 39.14°N | 100.39°E | 1360.80 | *Juncus effusus L* | 3.83 | 616.33 | 147.91 |
| $A_{13}$ | 39.22°N | 100.30°E | 1347.30 | *Scirpus triqueter L.* + *Echinochloa crusgalli (L.) Beauv.* | \ | \ | \ |
| $A_{14}$ | 39.29°N | 100.26°E | 1427.30 | *Heleocharis intersita Zinserl.* | 1011.64 | 186.73 | 150.54 |
| $A_{15}$ | 39.29°N | 100.26°E | 1337.00 | *Crypsis schoenoides (L.) Lam* | 1040.61 | 156.72 | 119.76 |
| $A_{16}$ | 39.29°N | 100.26°E | 1337.50 | *Juncellus serotinus (Rottb.) C.B. Clarke—Cyperus serotintts Rottb* | 1055.45 | 147.58 | 105.59 |
| $A_{17}$ | 39.29°N | 100.26°E | 1339.50 | *Herba Cirsii Setosi* | 1067.30 | 121.21 | 83.64 |
| $A_{18}$ | 39.29°N | 100.26°E | 1331.20 | *Inula britannica* | 1040.23 | 175.57 | 55.64 |
| $A_{19}$ | 39.29°N | 100.26°E | 1333.20 | *Scirpus validus Vahl* | 960.62 | 259.6 | 121.48 |
| $A_{20}$ | 39.31°N | 100.17°E | 1323.60 | *Phragmites australis (Cav.) Trin. ex Steud* | 448.09 | 227.28 | 379.55 |
| $A_{21}$ | 39.31°N | 100.17°E | 1324.90 | *Tamarix ramosissima Ledeb.* | 521.68 | 199.73 | 415.93 |
| $A_{22}$ | 39.22°N | 100.21°E | 1380.90 | *Potentilla anserina* | 94.25 | 1159 | 1872.57 |
| $A_{23}$ | 39.27°N | 100.04°E | 1338.10 | *Carex rigescens* | 8642.90 | 3370.47 | 7255.07 |
| $A_{24}$ | 39.33°N | 100.11°E | 1319.50 | *Phragmites Adans.* + *Calamagrostispseudophragmites* | 543.74 | 434.71 | 21.32 |
| $A_{25}$ | 39.36°N | 100.00°E | 1304.10 | *Acorus calamus* | 263.11 | 183.52 | 163.58 |
| $A_{26}$ | 39.40°N | 99.75°E | 1289.50 | *Inulasalsoloides (Turcz.) Ostrnf.* | 2181.62 | 437.61 | 8.76 |
| $A_{27}$ | 39.40°N | 99.75°E | 1288.20 | *Rumex crispus L.* | 1916.11 | 578.11 | 109.07 |
| $A_{28}$ | 39.40°N | 99.75°E | 1286.90 | *Scirpus planiculmis Fr. Schmidt* | 1897.85 | 372.19 | 37.39 |
| $A_{29}$ | 39.40°N | 99.75°E | 1286.30 | *Phragmites australis (Cav.) Trin. ex Steud* | 1888.59 | 363.48 | 47.29 |
| $A_{30}$ | 39.48°N | 99.65°E | 1274.00 | *Scirpus validus Vahl* | 2557.42 | 236.42 | 31.52 |
| $A_{31}$ | 39.59°N | 99.63°E | 1255.80 | *Glycyrrhiza uralensis Fisch.* | 1534.59 | 1000.8 | 18.40 |
| $A_{32}$ | 39.59°N | 99.63°E | 1255.10 | *Salicornia europaea* + *Phragmites australis (Cav.) Trin. ex Steud* | 1593.67 | 1079.88 | 55.42 |
| $A_{33}$ | 39.76°N | 99.48°E | 1226.60 | *Suaeda glauca (Bunge) Bunge* + *Agropyron cristatum (Linn.) Gaertn.* | 99.12 | 573.84 | 448.45 |
| $A_{34}$ | 39.76°N | 99.48°E | 1228.40 | *Alhagi sparsifolia Shap* + *Phragmites australis (Cav.) Trin. ex Steud* | 122.65 | 551.03 | 377.84 |
| $A_{35}$ | 39.75°N | 99.50°E | 1238.10 | *Salix wilhelmsiana* | 328.96 | 1676.86 | 34.55 |
| $A_{36}$ | 38.99°N | 100.41°E | 1410.30 | *Oxytropis glabra DC.* | 55.84 | 721.64 | 102.84 |
| $A_{37}$ | 38.99°N | 100.41°E | 1407.10 | *Echinochloa crusgalli var. mitis* | 26.27 | 707.91 | 118.11 |
| $A_{38}$ | 38.96°N | 100.44°E | 1416.30 | *Phragmites australis (Cav.) Trin. ex Steud* + *Agropyron cristatum (Linn.) Gaertn.* | 2429.19 | 646.45 | 14.97 |
| $A_{39}$ | 38.99°N | 100.48°E | 1401.30 | *Iris lactea* | 4541.15 | 629.22 | 172.75 |
| $A_{40}$ | 39.00°N | 100.47°E | 1397.70 | *Melilotus officinalis (L.) Lam.* + *Agropyron cristatum (Linn.) Gaertn.* | 3891.11 | 180.34 | 37.68 |
| $A_{41}$ | 38.99°N | 100.47°E | 1404.70 | *Phragmites australis (Cav.) Trin. ex Steud* | 3275.97 | 309.22 | 21.11 |
| $A_{42}$ | 38.97°N | 100.46°E | 1411.50 | *Phragmites australis (Cav.) Trin. ex Steud* + *Typha orientalis Presl.* | 3589.78 | 188.75 | 65.20 |

and lower herbaceous (lower than 60 cm) categories containing the following assemblages (Zhao et al., 2013), respectively: (1) *Elaeagnus angustifolia*, *Salix wilhelmsiana*, *Tamarix arceuthoides*, *Tamarix ramosissima*; (2) *Phragmites communis*, *Typha orientalis*, *Scirpus validus*, *Acorus calamus*, *Rumex crispus*; and (3) *Echinochloa crusgalli*, *Achnatherum splendens*, *Juncellus serotinus*, *Potentilla anserine*, *Scirpus planiculmis*, *Taraxacum officinale*, *Elymus dahuricus*, *Chenopodium glaucum*, and *Polygonum lapathifolium*, among others.

### 3.6. Statistical analysis of soil properties

Data were first checked using the one-sample Kolmogorov–Smirnov test to identify whether they were distributed in a nearly normal fashion. Then, the Independent-Samples Kruskal–Wallis test followed by the one-sample Kolmogorov–Smirnov test was used to explore whether changes in soil properties were the same across different LUCC categories. Before the test, a null hypothesis, that the distribution of soil properties was the same across LUCC categories, was set. If the results of the Independent-Samples Kruskal–Wallis Test indicated that difference between groups was significant ($P < 0.05$), we considered the null hypothesis to be rejected. Otherwise, we assumed that soil properties were the same across LUCC categories. Additionally, a scatter matrix was selected to explore differences in soil properties between various land use change and land cover types, and principal component analysis was applied to analyze the effects of LUCC on soil properties. The scatter matrix method, and principal component analysis were performed using SPSS 19.0, Pcord 5, and Canoco for Windows 4.5, respectively.

## 4. Results

### 4.1. The process of land use change

The middle reaches of the Heihe River produce a large percentage of all commodity grains produced by China, and arable land is the predominant land use type in the riparian zones of these reaches owing to the rich irrigation water (Fig. 3). Accordingly, the high density of human activity in the region plays a key role in land use change processes. In particular, land use change in the study area has manifested primarily as increases (decreases) in arable land (grasslands) throughout the study period (Fig. 3). Up to 309.02 $km^2$ of grassland was lost during the study period (Table 4), whereas arable land increased by 383.12 $km^2$ in 2010 alone owing to reclamation of grasslands, desert lands, and wetlands. Therefore, transformation of land use in the study area involved primarily the conversion from grasslands to arable land. In the past several decades, approximately 234.29 $km^2$ of grasslands was exploited to produce arable land, accounting for 61.16% of the growth of arable land. Major transitions also occurred between other land use types, including grassland–desert and wetland–grassland transitions. In fact, the transition matrix (Table 4) indicates that approximately 25.94 $km^2$ (11.12 $km^2$) of wetlands was converted into grasslands (desert). According to related researches in this region (Fu et al., 2014; Jiang et al., 2014; Li and Zhao, 2010), human activities are the main driving forces to the land use change. The reclamation of wetlands and grassland, water division policy and the construction land expansion all made that the land use in the riparian zone of the middle Heihe River basin has a great change in the past decades. Taking the transition of wetlands–desert as the example, it mostly is the result of the water allocation change and the overexploitation of groundwater resources. Wang et al. (2005) found that the groundwater recharge rate decreased by $2.168 \times 10^8$ $m^3$/a during the 1970s and 1980s, whereas the amount of groundwater storage has decreased by $0.545 \times 10^8$ $m^3$/a since 1986. As a result, water that can be used for wetland tended to be decreased. Especially, the water division made this situation more badly. Without the supplication of water, wetlands became smaller and fragmented, and then converted into desert land at last.

### 4.2. Soil properties for different land use change types

The direction of land use change has important effects on soil properties, with different land use transitions resulting in different soil properties. Based on values averaged over depths of 0–40 cm, soil moisture was found to be highest for arable–wetland transitions and lowest for wetland–desert transitions (Fig. 4).

The mean soil moisture content following arable–wetland transitions reached 22.07%, which is approximately nine times that for wetland–desert transitions. Conversely, soil bulk density was found to be lowest for arable–wetland transitions.

Average pH at depths of 0–40 cm was in the range of 8.59–9.13 and did not vary significantly with transition type, although the grassland–desert transition tended to be associated with higher pH values: these values were found to be 8.93, 9.24, and 9.22 for the 0–10, 10–20, and 20–40 cm layers, respectively (Fig. 4).

These results suggest the enhancement of alkalinity during the grassland–desert transition. No pronounced pH variability was observed for other transition types.

At depths of 0–40 cm, soil organic carbon storage varied significantly between different land use change types. In particular, mean soil organic carbon storage for the arable–wetland transition was 11.45 g/kg, which is much greater than that for the arable–grassland transition. Similarly, the mean storage for land that had undergone a grassland–desert (grassland–arable) transition was found to be only 3.15 g/kg (7.29 g/kg). Significant variations in mean soil organic carbon storage were also found between different land use transitions for individual soil layers (Fig. 4).

For example, at depths of 10–20 cm, the mean storage for land that had undergone arable–wetland transition was found to be 12.19 g/kg, yet that for the arable–grassland transition was only 7.28 g/kg. Furthermore, the mean storage for land that had undergone grassland–arable (grassland–desert) transition was found to be 6.26 (3.16), 8.13 (2.73), and 7.29 (3.15) g/kg for the layers at 0–10, 10–20, and 20–40 cm, respectively (Fig. 4). Wetlands are known to act as huge pools of organic carbon globally (Zhang et al., 2011). Accordingly, transitions from wetlands to other land use types typically result in considerable loss of soil organic carbon. For depths of 0–10 and 10–20 cm, we found soil organic carbon storage to be greater for areas that had undergone wetland–arable transitions than for those experiencing wetland–desert transitions; however, we found the opposite relationship for depths of 20–40 cm. This can be attributed primarily to deep plowing and careful cultivation during wetland reclamation for use as arable land: both of these processes tend to promote the uniform distribution of soil organic carbon with depth. Conversely, the transition from wetlands to desert typically promotes changes in soil organic carbon storage only at the soil surface.

We also investigated the influence of land use change on nutrient-related parameters such as total and available nitrogen, phosphorous, and potassium. For the 0–10, 10–20, and 20–40 cm depths, we found total potassium storage to be 10.18, 9.73, and 10.49 g/kg, respectively, in regions that had undergone a transition from wetlands to arable land; these values are much greater than those found for any other land use transitions and were complemented by abundant stored total nitrogen and phosphorous and available phosphorous, nitrogen, and potassium. These characteristics indicate indirectly that wetlands are an ideal resource for agricultural development. In contrast, the wetland–desert transition is associated with loss of soil nutrients. We determined average storage of total (available) nitrogen, phosphorous, and potassium to be only 1.11 (0.0176), 1.98 (0.0398), and 6.16 (0.0058) g/kg, respectively, for regions that had undergone such a transition (Fig. 4).

These values are lower than those obtained for all other transition types, suggesting that the degradation of wetlands leads to reductions in soil quality. Similar degradations in soil quality were also found in other conversions, although to a lesser degree. Of these degradations, that associated with the arable–wetland transition was the most

**Table 4**
The transition matrix of land use during 1975–2010 (units: $km^2$).

| 1975 | 2010 | | | | | | |
|---|---|---|---|---|---|---|---|
| | Construction lands | Desert land | Arable land | Forest land | Grassland | Wetland | Total |
| Construction land | 117.91 | 0.01 | 4.95 | 0.04 | 0.21 | 0.06 | 123.16 |
| Desert land | 2.64 | 220.68 | 63.08 | 16.28 | 69.23 | 10.30 | 382.20 |
| Arable land | 22.27 | 5.01 | 1105.31 | 3.20 | 15.49 | 4.48 | 1155.76 |
| Forest land | 0.39 | 6.31 | 29.70 | 60.78 | 15.15 | 0.15 | 112.48 |
| Grassland | 3.16 | 44.92 | 234.29 | 18.84 | 124.86 | 7.81 | 433.89 |
| Wetland | 0.33 | 11.12 | 51.10 | 1.56 | 25.94 | 80.92 | 170.98 |
| Total | 146.71 | 288.04 | 1488.43 | 100.70 | 250.88 | 103.71 | 2378.47 |

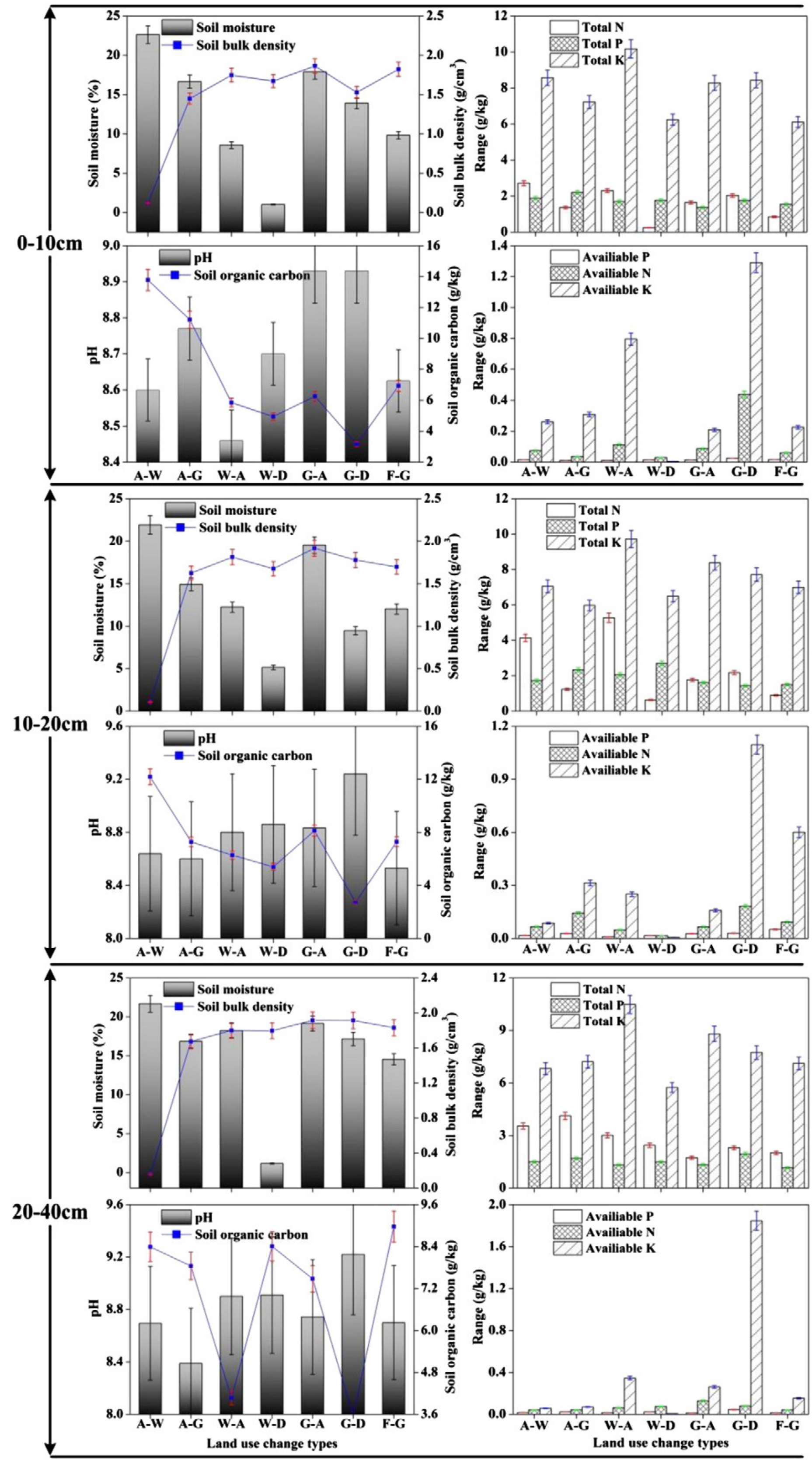


**Fig. 4.** Soil properties at various depths for regions that have undergone different land use transition types (n = 16). Values are given as means ± S.E. Abbreviations on horizontal axes refer to land use transition types, where A, W, G, D, and F refer to arable lands, wetlands, grasslands, desert lands, and forest lands, respectively.

**Table 5**
Soil properties at different depths for different land covers.

| Vegetation type | Depth (cm) | Soil moisture (%) | Soil bulk density ($g/cm^3$) | pH | Soil organic carbon (g/kg) | Total nitrogen (g/kg) | Total phosphorous (g/kg) | Total potassium (g/kg) | Available phosphorous (g/kg) | Available nitrogen (g/kg) | Available potassium (g/kg) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Woody vegetation | 0–10 | 15.54 ± 5.31 | 1.78 ± 0.05 | 8.70 ± 0.21 | 6.27 ± 1.18 | 2.11 ± 1.13 | 1.91 ± 0.34 | 6.68 ± 0.49 | 0.015 ± 0.005 | 0.06 ± 0.02 | 0.25 ± 0.10 |
| | 10–20 | 17.58 ± 5.20 | 1.72 ± 0.03 | 8.63 ± 0.17 | 9.28 ± 2.07 | 1.02 ± 0.32 | 1.62 ± 0.18 | 7.18 ± 0.27 | 0.03 ± 0.02 | 0.07 ± 0.02 | 0.34 ± 0.28 |
| | 20–40 | 19.73 ± 5.44 | 1.75 ± 0.13 | 8.72 ± 0.04 | 6.47 ± 2.52 | 1.24 ± 0.83 | 1.57 ± 0.27 | 7.06 ± 0.69 | 0.016 ± 0.003 | 0.04 ± 0.01 | 0.09 ± 0.05 |
| Tall herbaceous vegetation | 0–10 | 18.30 ± 1.91 | 1.64 ± 0.05 | 8.70 ± 0.08 | 12.54 ± 1.72 | 2.57 ± 0.71 | 2.00 ± 0.18 | 8.44 ± 0.85 | 0.019 ± 0.003 | 0.14 ± 0.04 | 0.33 ± 0.13 |
| | 10–20 | 17.76 ± 2.08 | 1.71 ± 0.06 | 8.79 ± 0.09 | 9.63 ± 1.47 | 3.20 ± 1.09 | 1.84 ± 0.22 | 8.20 ± 0.80 | 0.018 ± 0.003 | 0.17 ± 0.05 | 0.26 ± 0.11 |
| | 20–40 | 18.68 ± 2.58 | 1.82 ± 0.05 | 8.70 ± 0.10 | 8.36 ± 1.25 | 2.35 ± 0.50 | 2.02 ± 0.23 | 6.98 ± 0.83 | 0.02 ± 0.005 | 0.10 ± 0.02 | 0.33 ± 0.19 |
| Low grass vegetation | 0–10 | 17.64 ± 1.51 | 1.73 ± 0.04 | 8.79 ± 0.06 | 9.89 ± 1.23 | 1.98 ± 0.25 | 1.93 ± 0.12 | 7.53 ± 0.51 | 0.018 ± 0.002 | 0.20 ± 0.07 | 0.53 ± 0.13 |
| | 10–20 | 16.91 ± 1.41 | 1.78 ± 0.03 | 8.79 ± 0.06 | 7.58 ± 0.66 | 2.05 ± 0.29 | 1.87 ± 0.13 | 7.40 ± 0.45 | 0.02 ± 0.003 | 0.08 ± 0.01 | 0.27 ± 0.07 |
| | 20–40 | 19.18 ± 1.17 | 1.79 ± 0.03 | 8.77 ± 0.06 | 8.18 ± 0.66 | 3.06 ± 0.41 | 1.86 ± 0.13 | 8.07 ± 0.38 | 0.02 ± 0.002 | 0.10 ± 0.02 | 0.24 ± 0.05 |

n = 42 (5, 10, and 27 samples for woody, tall herbaceous, and low grass vegetation, respectively). Values are given as mean ± S.E.

pronounced, followed by the arable–grassland, grassland–arable, and grassland–desert transitions.

### 4.3. Variation in soil properties with land cover

Table 5 presents the soil properties at different depths for different land covers. We found significant differences in soil moisture between areas of woody, tall herbaceous, and lower herbaceous vegetation, with soil moisture across all land cover types in the range of 15.54–19.73% (Table 5). Moreover, soil moisture content was higher in the 20–40 cm layer than at other depths. This distribution was particularly pronounced under woody vegetation, where soil moisture content reached 19.73%. Soil bulk density was also highest for the 20–40 cm layer, such that soil moisture and bulk density exhibited a positive relationship over this interval. Conversely, soil moisture and bulk density exhibited a negative relationship (i.e., high moisture content but low bulk density) for the 0–10 and 10–20 cm intervals.

All land cover types in the study area were found to exhibit alkaline pH values, in accordance with the characteristics typical of arid environments. In particular, soil covered with low grass vegetation typically exhibited higher pH than that covered by other land cover types (Table 5), although this difference was not found to be particularly significant in general.

Tall herbaceous vegetation and low grass vegetation were associated with higher soil organic carbon content than woody vegetation (Table 5). In particular, the soil organic carbon content was found to be highest under tall herbaceous vegetation, reaching 12.54 (±1.72) g/kg in the 0–10 cm layer. Variations in soil organic carbon distribution with depth were also found. For example, soil organic carbon was typically concentrated in the 0–10 cm (10–20 cm) layer for tall herbaceous and low grass vegetation (woody vegetation). This distribution suggests that land cover change involving the transition from herbaceous to woody plants has the potential to result in decreases in soil organic carbon in arid zones.

We selected several parameters (total and available nitrogen, phosphorous, and potassium) for further investigation to compare the effects of different land cover types and change on soil nutrients. Our results (Table 5) demonstrate that nutrients are most abundant in soil underlying tall herbaceous vegetation, with total nitrogen, phosphorous, and potassium in the 0–10 cm layer reaching up to 2.57 (±0.71), 2.00 (±0.18), and 8.44 (±0.85) g/kg, respectively. These values are significantly higher than those for soil underlying low grass vegetation and woody vegetation. Moreover, although nutrients in soils underlying woody vegetation are typically less abundant than those in soils under tall herbaceous vegetation, the soils supporting woody vegetation typically contain abundant available phosphorous, nitrogen, and potassium. For example, available potassium in the 0–10 cm layer of such soils reached up to 0.53 (±0.13) g/kg.

### 4.4. Effects of LUCC on soil properties

The null hypothesis of Independent-Samples Kruskal–Wallis Test assumed that soil properties were the same across LUCC categories. If the *P* value was less than 0.05, we rejected this hypothesis; otherwise, the hypothesis was retained. Our results (Tables 6, 7) show that the distributions of soil moisture, soil bulk density, pH, soil organic carbon, available K, and total K were not constant across land use change categories. However, the indices of other soil properties suggested that the null hypothesis be retained. The distributions of soil organic carbon, available P, available K, total P, and total K were not found to be constant across categories of land cover, whereas soil moisture, soil bulk density, pH, available N, and total N values suggested that the null hypothesis be retained.

The scatter matrix (Fig. 5-a) indicates that the soil properties of the wetland–desert transition were not correlated significantly with other land use change types, although correlations were found for the other

**Table 6**
Independent-Samples Kruskal–Wallis Test of different land covers.

| No. | Null hypothesis | Test | Sig. | Decision |
|---|---|---|---|---|
| 1 | The distribution of soil moisture is the same across categories of land cover | Independent-Samples Kruskal-Wallis Test | 0.601 | Retain the null hypothesis |
| 2 | The distribution of soil bulk density is the same across categories of land cover | Independent-Samples Kruskal-Wallis Test | 0.179 | Retain the null hypothesis |
| 3 | The distribution of pH is the same across categories of land cover | Independent-Samples Kruskal-Wallis Test | 0.268 | Retain the null hypothesis |
| 4 | The distribution of soil organic carbon is the same across categories of land cover | Independent-Samples Kruskal-Wallis Test | 0.044 | Reject the null hypothesis |
| 5 | The distribution of available P is the same across categories of land cover | Independent-Samples Kruskal-Wallis Test | 0.019 | Reject the null hypothesis |
| 6 | The distribution of available N is the same across categories of land cover | Independent-Samples Kruskal-Wallis Test | 0.503 | Retain the null hypothesis |
| 7 | The distribution of available K is the same across categories of land cover | Independent-Samples Kruskal-Wallis Test | 0.044 | Reject the null hypothesis |
| 8 | The distribution of total N is the same across categories of land cover | Independent-Samples Kruskal-Wallis Test | 0.591 | Retain the null hypothesis |
| 9 | The distribution of total P is the same across categories of land cover | Independent-Samples Kruskal-Wallis Test | 0.012 | Reject the null hypothesis |
| 10 | The distribution of total K is the same across categories of land cover | Independent-Samples Kruskal-Wallis Test | 0.036 | Reject the null hypothesis |

Asymptotic significances are displayed. The significance level is 0.05.

change types. This suggests that the soil properties associated with the wetland–desert transition are considerably different to those associated with other transitions. In contrast, the significant correlations found for the other change types suggest that the soil properties studied exhibit no significant differences between various land covers.

From the principal component analysis (Fig. 6), we found that soil moisture was well explained by LUCC; other parameters, such as soil organic carbon, pH, and total K, were also closely related to LUCC. However, changes in soil bulk density, available P, available N, available K, total P, and total N were proved to exhibit opposite correlations with LUCC. Furthermore, changes in pH and total K can be explained well by the wetland–desert transition, and changes in soil organic carbon are associated primarily with the following transitions: wetlands–arable, forest–grasslands, and grasslands–desert. Conversely, the observed variations in soil moisture can be attributed primarily to land cover rather than land use changes.

**Table 7**
Independent-Samples Kruskal–Wallis Test of different land use change types.

| No. | Null hypothesis | Test | Sig. | Decision |
|---|---|---|---|---|
| 1 | The distribution of soil moisture is the same across categories of land use change | Independent-Samples Kruskal-Wallis Test | 0.011 | Reject the null hypothesis |
| 2 | The distribution of soil bulk density is the same across categories of land use change | Independent-Samples Kruskal-Wallis Test | 0.016 | Reject the null hypothesis |
| 3 | The distribution of pH is the same across categories of land use change | Independent-Samples Kruskal-Wallis Test | 0.040 | Reject the null hypothesis |
| 4 | The distribution of soil organic carbon is the same across categories of land use change | Independent-Samples Kruskal-Wallis Test | 0.024 | Reject the null hypothesis |
| 5 | The distribution of available P is the same across categories of land use change | Independent-Samples Kruskal-Wallis Test | 0.251 | Retain the null hypothesis |
| 6 | The distribution of available N is the same across categories of land use change | Independent-Samples Kruskal-Wallis Test | 0.223 | Retain the null hypothesis |
| 7 | The distribution of available K is the same across categories of land use change | Independent-Samples Kruskal-Wallis Test | 0.020 | Reject the null hypothesis |
| 8 | The distribution of total N is the same across categories of land use change | Independent-Samples Kruskal-Wallis Test | 0.069 | Retain the null hypothesis |
| 9 | The distribution of total P is the same across categories of land use change | Independent-Samples Kruskal-Wallis Test | 0.224 | Retain the null hypothesis |
| 10 | The distribution of total K is the same across categories of land cover | Independent-Samples Kruskal-WallisTest | 0.015 | Reject the null hypothesis |

Asymptotic significances are displayed. The significance level is 0.05.

## 5. Discussion

### 5.1. LUCC results in significant soil moisture changes

Soil moisture content plays a key role in energy exchange between the various components of ecosystems (Venkatesh et al., 2011; Wang et al., 2013), and the arid climate of the middle reaches of the Heihe River has likely exacerbated the loss of soil moisture during LUCC processes. Figs. 4 and 6, and Table 5 illustrate changes in soil moisture content for different LUCC parameters and demonstrate significant differences between LUCC types.

Table 6 further demonstrates that the distribution of soil moisture remains constant across land cover categories. Moreover, Table 7 exhibits a significant positive correlation between soil moisture and land use change, suggesting that the distribution of soil moisture does not remain constant across land use change categories. This is in stark contrast to the results of Qiu et al. (2001), who attributed changes in soil moisture primarily to topography rather than land use. Based on our results, it appears that the land use transition from wetlands to desert represents one of the major pathways for soil moisture loss in the study area.

Our results also demonstrate that the soil moisture content for grassland vegetation was greater than that for woody vegetation, particularly for tall herbaceous vegetation, which is associated with the highest soil moisture contents in our study area. However, we found the difference in soil moisture between land cover types to be insignificant (Figs. 4, 5, Table 5). This is consistent with the results of Wang et al. (2013), who investigated the soil moisture content of different land cover types in a semiarid environment and found soil moisture to decrease in the following order: crops, grass, subshrub, tree, and shrub. Moreover, land cover can account for changes in soil moisture better than land use change, according to the results of our principal component analysis (Fig. 6). This is consistent with the fact that vegetation typically acts as an indicator of the abundance of water in arid zones.

### 5.2. Effects of LUCC on soil organic carbon

As a key factor in the balance of soil organic carbon, LUCC can both increase and decrease the storage of soil organic carbon (Boix-Fayos et al., 2009; Cheng et al., 2013; Poeplau and Don, 2013). Moreover, because the present study was conducted in an arid region, the effects of LUCC on soil organic carbon are likely to be particularly pronounced (Su et al., 2010).

We found the transition from arable land to wetlands to be associated with significant increases in the storage of soil organic carbon (Figs. 4, 6). Conversely, the transition from grasslands to desert was associated with considerable decreases in soil organic carbon (Figs. 4, 6). We attribute this primarily to the role of wetlands as a key carbon sink globally (Mer and Roger, 2001; Wang et al., 2011). The incomplete factorization of soil organic matter and carbon fixation by wetlands act to increase soil organic carbon with respect to other land use types. Therefore, corresponding decreases in soil organic carbon will occur where wetlands are converted into other land use types. Furthermore, we found areas that have undergone an arable–grassland transition to exhibit the second highest soil carbon content; this is in agreement with the results of Su (2007) and Wu et al. (2003) for soil carbon sequestration in northwestern China. Thus, although land use change can act to increase soil organic carbon, land subjected to land use change typically experiences soil carbon loss. For this reason, irrational land use change should be discouraged in arid riparian zones (Qiu et al., 2012).

We also found that soil organic carbon tends to vary with land cover type, and the differences between various land covers were significant, with correlations decreasing in the following order: tall herbaceous vegetation, low grass vegetation, and woody vegetation. Thus, soil organic carbon storage was highest for tall herbaceous vegetation and lowest for woody vegetation (Table 5). This result is consistent with the results presented by Wang et al. (2006), who concluded that land covered with

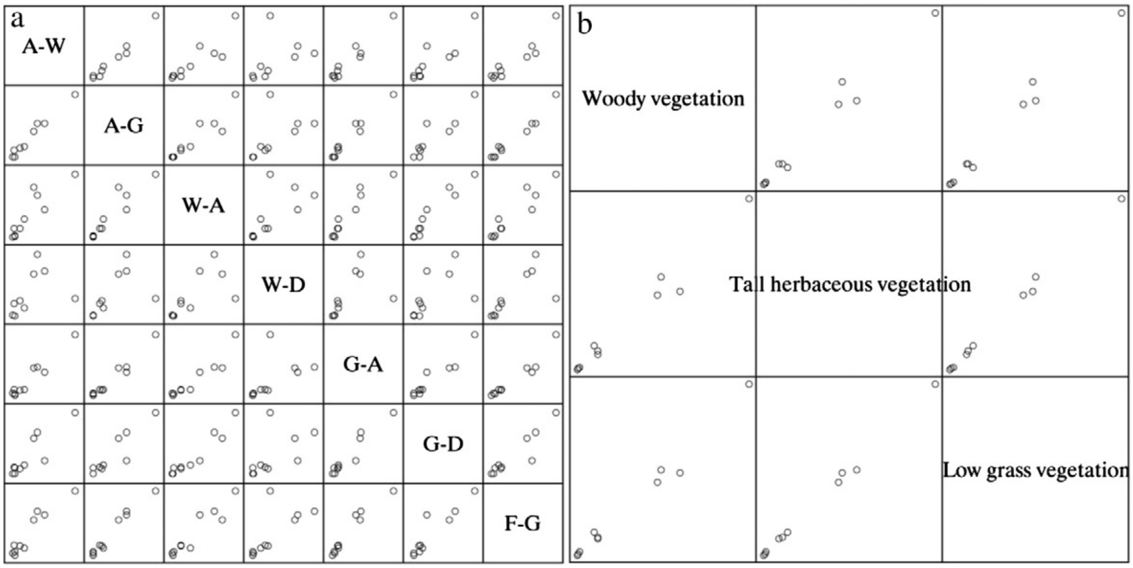


**Fig. 5.** Scatter matrix for the soil properties of different land use change (a) and land cover types (b).

integrated grass and sparse brushwood typically exhibits the highest soil organic carbon contents.

### 5.3. Effects of LUCC on soil nutrient elements

In the arid regions of China, soil nutrients such as nitrogen, phosphorous, and potassium are key components of soil quality and agricultural production (Su, 2007; Zhang et al., 2005; Zhang et al., 2009). In particular, soil nutrients in the middle reaches of the Heihe River (where a large percentage of northwestern China's marketable grains are produced) play a key role in maintaining grain output over time. Our results demonstrate that sharp changes in land use and land cover have had considerable finite effects on soil nutrients in this region (Fig. 6). Although total K was found to exhibit a significantly positive correlation with LUCC, the values of other parameters studied, such as total P and N and available P, N, and K, could not be explained well by LUCC. Therefore, LUCC cannot be considered as the predominant factor controlling soil nutrients in the study area. In fact, the riparian zone of the middle Heihe River basin has always been the most important farming area in China, and thousands of years of cultivation in this region have made the soil more fertile and increased the contents of soil nutrients. In this context, manual interventions of farmers to address soil quality will be a key factor in promoting changes in soil nutrient contents.

As suggested by Kong et al. (2006) and Wei et al. (2013), we also believe that some land use changes can act to improve soil nutrient contents, such as the conversion of wetlands to arable land (rather than desert) or arable land to wetlands (rather than grasslands). Similarly, Wei et al. (2013) demonstrated that soil nutrient contents were higher following the conversion of wetlands to drylands than after the transition from wetlands/drylands to paddy lands. Moreover, according to the results of our principal component analysis (Fig. 6), changes in soil nutrients also exhibit significant opposite correlations with LUCC, although the proportions of their variations that can be explained by LUCC are typically lower than that those explained by the above-mentioned conversions (Fig. 6). Therefore, further research is required to determine the mechanisms underlying such changes.

### 5.4. Recommendations of the future land use/land cover change

The riparian zone is the most key part of the desert oasis in the arid region. It is also the most vulnerable ecosystem and needs to be protected urgently. An appropriate land use will contribute more to the sustainable development. In the riparian zone of the middle Heihe

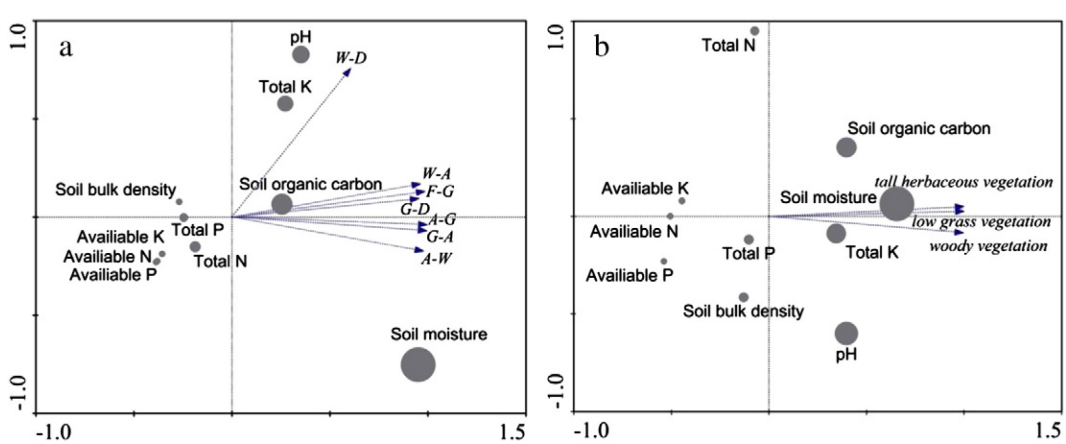


**Fig. 6.** Principal component analysis for the soil properties of different land use change (a) and land cover types (b), where A, W, G, D, and F refer to arable lands, wetlands, grasslands, desert lands, and forest lands, respectively. (The size of the circles indicates the proportions that variations can be explained by the ordination model. The vertical distances from the circles to the arrow lines indicate theirs correlations. Short distance shows a close correlativity. Conversely, the correlativity is not significant. Furthermore, the perpendicular lines between the circles and arrow lines locate in the backward extension lines mean an opposite correlation. Otherwise, it means a positive correlation.).

River basin, the enlargement of farmland and desert land made that the structure of land use tend to be unreasonable and unhealthy, characterized as the wetland landscape fragmentation (Jiang et al., 2014; Li and Zhao, 2010) and the reduction of the forest and grassland (Fig. 3).

Since the wetland, forest and grassland are the ecological barrier and existing infrastructure of the oasis, the protection measures of these ecological land use types should be accepted. Nature reserve for wetland, water source, forest and so on should be constructed as soon as possible. Moreover, the local government needs to appropriately restrain the scale of farmland reclamation and the enlargement of the construction land. Besides the measures illustrated above, greening transformation also should be started to change the land cover state of the desert lands. Especially the ecological restoration of the wetland is urgently needed to prevent the wetlands from fragmenting and degradation. Generally speaking, the future policy of land use in the study area is the stabilization of farmland and the enlargement of wetland.

## 6. Conclusions

The present study indicates that soil properties in the riparian zones of desert oasis are affected significantly by LUCC. In particular, our results demonstrate that changes in soil moisture and soil organic carbon are correlated significantly with LUCC processes. Our selected methods have shown that the distributions of soil moisture, soil bulk density, pH, soil organic carbon, available K, and total K are not constant across land use change categories. Similarly, the distributions of soil organic carbon, available P, available K, total P, and total K are not constant across land cover categories. In particular, soil moisture variation can be well explained by LUCC and is attributed primarily to land cover changes rather than land use changes. The observed changes in soil organic carbon, pH, and total K were also found to be closely related to LUCC and can be explained by wetland–arable, forest–grassland, grassland–desert, and wetland–desert transitions, respectively. However, the observed changes in soil nutrients, such as total P and N and available P, N, and K, could not be explained well by LUCC. Rather, we attribute these changes in soil nutrients primarily to human agricultural activity. Generally speaking, we believe that this study has well illustrated the impacts of LUCC on soil properties in the riparian zones of desert oasis by coupling experimental data with remote sensing data. However, the lack of soil data spanning long time periods makes it difficult for us to analyze the effects of LUCC on soil properties in terms of spatiotemporal variability. Therefore, further research is required to determine the mechanisms underlying such changes.

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