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Land exploitation resulting in soil salinization in a desert–oasis ecotone

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ABSTRACT

Understanding the process of agricultural land expansion and its impact on soil properties is crucial for land management and environmental health. A desert–oasis ecotone is typically located between an oasis at the lower reach of inland rivers and neighboring desert in arid regions, and acts as an interactive zone between irrigated farmland and the natural desert ecosystem. The arid region of northwest China has experienced dramatic land exploitation since the 1960s and soil salinization has been a serious environmental problem ever since. The objective of this study was to determine the relationship between land exploitation and soil salt accumulation in the topsoil. A typical desert–oasis ecotone, the Fubei region at the lower reach of the Sangong River catchment in arid northwest China, was used as a case study. The results revealed the following: (1) overexploitation of land resources has been astonishing since 1960, with >40% of the area becoming irrigated farmland. There was frequent transition of land-use types from one to another, with about 38% of the area experiencing transitional change during 1982–2009. (2) Comparing soil salt content with land use during 1982–2009 showed an expanding area of soil salinity and an increased degree of salinity in all land-use types. The area with soil salt content > 20 g/kg increased by 16.4%, while the area with soil salt content of 5–10 g/kg decreased by 42% during 1982–2009. In addition, the amount of overall soil salt accumulation was about 21.6×10^{10} g in the study area during 1982–2009, and soil salt accumulation per unit area increased by 60%, with salt accumulation in farmland, grassland and saline-alkali land higher than for other land-use types ($p<0.05$). (3) The dramatic salt accumulation was a result of agricultural land exploitation that requires irrigation, and this directly caused a rising groundwater table, and then higher evaporation led to soil salinization. Collectively, the results indicate that overexploitation of land resources had large and prolonged effects on soil salinization, which is a lesson to be learned for integrated land management in similar ecotones in arid zones.

1. Introduction

Soil salinization is a long-standing environmental problem in the world, especially in arid or semiarid regions. As a recurring issue concerning environmental health, it has long received a good deal of attention (Bennett et al., 2009; Farifteh et al., 2006; Jordán et al., 2003; Qadir et al., 2000; Thomas and Middleton, 1993). Soil salinization is a major environmental threat affecting soil properties and reducing soil productivity (Tilman et al., 2002). Thus, more effort is needed to improve productivity as more land becomes salinized (Rengasamy, 2006). It has been estimated that approximately 955 M ha of land is salt-affected worldwide, while secondary salinization affects about 80 M ha of land in arid and semiarid regions (Ghassemi et al., 1995). Over the past 50 years, humans have changed ecosystems more rapidly and more extensively than at any other time in human history (Bennett and Balvaera, 2007), and this mainly appears in the changing pattern of land use. About 13 M ha of land is converted to agricultural use each year. Moreover, rapid land-use change has been associated with deleterious environmental outcomes in the socioeconomic processes of human activities (Liu and Diamond, 2005; Versace et al., 2008). The conversion of natural landscapes such as grasslands and forests to agricultural land has a dramatic impact on soil properties, especially in accelerating soil salinization (Houk et al., 2006; Qureshi et al., 2008). In arid areas, soil salinization is mainly caused by rising water tables as a consequence of reduced evapotranspiration following changes in land-use pattern, typically land exploitation for agriculture (Williams et al., 1997). Many exploited lands increase groundwater recharge and bring soil-stored soluble salts to the surface (Ritzema et al., 2008). In previous studies on land-use changes, limited attention had been paid to soil salinization following these changes. Therefore, monitoring of soil salinity must be urgently implemented in order to evaluate the progression of salinity hazards and the effectiveness of remediation strategies (Douaik et al., 2007; Douaoui et al., 2006; Martínez-Sánchez et al., 2011).

Desert–oasis ecotones are typically located between an oasis at the lower reach of inland rivers and the neighboring desert in an arid area, and act as an interactive zone between irrigated farmland and the natural desert ecosystem. The ecotone is a narrow belt

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between desert and oasis and yet may play a prominent ecological role that far exceeds their physical extent: such as ensuring oasis ecological security and maintaining oasis internal stability. At landscape scales, desert–oasis ecotones function as a boundary and corridor between desert and oasis ecosystems, controlling the flux of energy and nutrients as well as being a biotic exchange between desert and oasis (Wang et al., 2007a). Hence, the ecotone is likely to be very sensitive to human activities such as land exploitation (Metzger et al., 2006), with soil salinization a frequently observed phenomenon of land degradation (Wang et al., 2008a). Over recent decades, soil salinization induced by human activities has been severe, which generally threatens environment health and sustainable development in the region. Thus, assessments of anthropogenic impacts on salinization are urgently needed to develop sound land-use policies and planning actions for integrated land management (Zhang et al., 2011).

Oases are a unique non-zonal landscape in arid regions that are created by exploitation of water resources from inland river basins (Li et al., 2007). Oasification and desertification are two complementary processes in the course of oasis evolution and often occur simultaneously in a desert–oasis ecotone (Zhang et al., 2003). Farmland is the most important part of an oasis land in arid zones, and their dynamic changes are a key issue affecting sustainable development of oases (Wahap et al., 2004). Land-use changes such as farmland exploitation affect the physical, chemical and biological processes of soil, and so alter soil properties (Sveistrup et al., 2005) such as the degree of soil salinity.

Salt accumulation in soil is an important factor threatening agricultural safety and regional stability in oases. Maintaining stability in a desert–oasis ecotone could effectively prevent land degradation/salinization in the oasis. Many studies have been conducted in arid environments to assess and monitor land-use change (Lioubimtseva et al., 2005; Luo et al., 2008) or soil salinization processes (Chen et al., 2010; Sheng et al., 2010). However, understanding the process of agricultural land expansion by human activity and its influence on soil properties is crucial for land management and environmental health (Bennett et al., 2010). The expansion of agriculture is posited as one of the main dynamics of land-use change globally, and understanding these processes is important for policy as well as academic concerns (David et al., 2001; McConnell et al., 2004). Therefore, quantifying the spatial and temporal changes in land use and soil salinization would allow us to identify the locations and extent of soil salinization during the course of agricultural land expansion. The objective of this study was to determine the influence of land-use change on expansion of salinized soils.

2. Material and methods

2.1. Study area

In the past few years, we have conducted a systematic study on a typical desert–oasis ecotone—Fubei in Xinjiang Province, northwest China—in an effort to provide a scientific basis for understanding land-use dynamics and its influence on soil salinization in a desert–oasis ecotone. The Fubei region (87°47′30″–88°01′15″E and 44°17′30″–44°22′30″N), a typical desert–oasis ecotone in the arid zone of northwestern China, is located at the lower reaches of the Sangong River catchment (Fig. 1). The Sangong River originates from the north slope of the Tianshan Mountains. The catchment covers the mountainous region in the south, the oasis region in the middle and the desert region in the north, which forms a mountain–oasis–desert landscape pattern typical of the arid region of northwest China. The south part of the oasis region is farmland with thousands of years of history; however, in the north, farmland is less than 60 years old. A reservoir was constructed in the 1970s at the end of the Sangong River to supply water for Fubei's agricultural irrigation. These hydrological and land-use changes have promoted soil salinization and land degradation in the entire lower reaches of the Sangong River catchment. The area of saline land has significantly increased in recent decades (Wang et al., 2008b).

The Fubei region covers an area of approximately 159 km^2, and has a population of 12,000. Dimensions of the region are 19.2 and 8.8 km for north–south and east–west, respectively. The topography is generally flat with a slope of 0.17% or less downwards from southeast to northwest. The elevation range of the area is 454.3–485.4 m. The climate is an arid continental climate with annual precipitation of 163 mm, and range in annual pan evaporation of 1780–2460 mm. Owing to the high evaporative demand in the lower reaches of the Sangong River

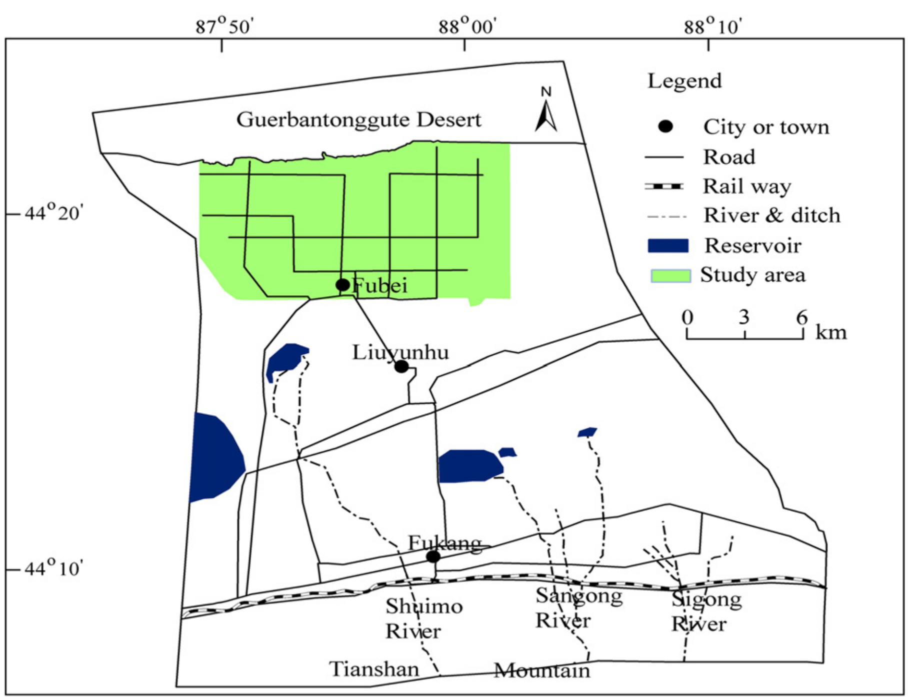

Fig. 1. Location map of the Fubei region in the Sangong River catchment, northwest China.

catchment, which is a plain with no surface runoff and slow groundwater flow rate, saline land is widely distributed, and soil salt accumulation (SSA) naturally occurs (Wang et al., 2008a). The main soil type according to FAO/UNESCO (1990) is Solonchak, covering approximately 37% of the study area, and less frequent soil types include Haplic calcisols and Aquert on substrates of varied fertility. Natural vegetation of the Fubei region is characterized by different types of xeric or halophyte communities, dominated by desert shrubs *Tamarix ramosissima*, *Haloxylon ammodendron* and *Reaumuria soongorica* scrubs. Crops in the oasis include cotton, wheat, hops, grapes and corn.

2.2. Data collection and analysis

Land-use change was obtained by a land-use map at 1:10,000 scale for 1982 and 2009 (Fubei Farm Land Resources Administration, 1982, 2009). The area measurements of the maps in the study were made using the statistical functions of GIS according to details of methodology and procedures (Bocco et al., 2001). The land-use maps of the study area were compiled based on the land-use map using GIS software ArcView 3.2a (Environmental Systems Research Institute Inc., USA). The study area can be divided into six land-use types: farmland, residential area, shrub land, planted forest, grassland and saline-alkali land in accordance with properties such as landform, landscape type and dominant plant species. No features such as roads or undeveloped desert were included (Fig. 2).

Soil salt content data for 1982 and 2009 were mainly from two sources: 147 soil samples measured in 2009 by the authors, and the other from Fubei Farm Land Resources Administration (1982, 2009) of 86 soil samples. The sample depth was 0–20 cm following the standard of the national soil survey, and sampling locations were recorded using GPS. The spatial distribution of soil salinity was determined by soil salt content data, elaborated using geostatistical methods as described in several studies (e.g. Campbell, 1978; Chien et al., 1997; Gao et al., 2010; Herrero and Pérez-Coveta, 2005; Yost et al., 1982). One important contribution of geostatistics is providing a map of the probability of soil salinity values (Castrignano et al., 2002). A semivariogram is a basic tool of geostatistics representing the spatial dependence of each point on its neighbor (Goovaerts, 1999). Its general form is as follows:

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} [z(x_i) - z(x_i + h)]^2 \quad (1)$$

where $\gamma(h)$ is a semivariogram, $z(x_i)$ is the measured sample value at point x_i, $z(x_i+h)$ is a measured sample at point x_i+h, and $N(h)$ is the number of pairs separated by lag h.

A GS$^+$ 5.3.2 program (Robertson, 2000) designed by Gamma Design Software was used to calculate $\gamma(h)$ and theoretical model parameters of $\gamma(h)$ for soil salt content data. This allowed calculation of the interpolated value (kriging). The ordinary kriging estimator, $z(x_0)$, of an unsampled site is a linear sum of weighted observations within a neighborhood as an optimal method of spatial prediction:

$$z(x_0) = \sum_{i=1}^{n} \lambda_i z(x_i) \quad (2)$$

where $z(x_0)$ is the value to be estimated at the location x_0, $z(x_i)$ is the known value at the sampling site x_i, and n is the number of sites within the search neighborhood used for the estimation; n is based on the size of the moving window and is defined by the user.

The mean error (ME) and the root mean square error (RMSE) were employed to assess the effectiveness of ordinary kriging. The ME should be close to zero and the RMSE should be close to one, suggesting unbiased prediction and its precision between estimated and actual values, and indicating that the predicted map is credible. An ordinary kriging method in ARCGIS's geostatistical analysis module was used to make soil salinity maps for two periods (1982 and 2009); then digital soil salt content maps were combined with region land-use types to analyze the relationship of temporal–spatial soil salt content with land-use types at a region scale.

We developed an equation to quantify and therefore characterize the SSA state in different land-use types based on distribution of degree of soil salt content in land-use type. For an individual land-use type with k patches, the equation below was used to calculate SSA (g):

$$SSA_d = \sum_{i=1}^{k} SSA_i = \sum_{i=1}^{k} \rho_i \times S_i \times D_i \times (1 - V_i) \times A_i \quad (3)$$

$$SSA_i = \rho_i \times S_i \times D_i \times (1 - V_i) \times A_i \quad (4)$$

where k is the number of patches, ρ_i is the bulk density (Mg m^{-3}), S_i is the distribution area of soil salt content (m^2) in degree of soil salt content i, D_i is the thickness of soil layer (m), V_i is the volume fraction of fragments > 2 mm, and A_i is the mean of the soil salt content (g kg^{-1}). Soil bulk density data in 1982 (Fubei Farm Land Resources Administration, 1982, 2009) and soil bulk density in the same situation and land-use type were determined using a soil corer (stainless steel cylinder of 100 cm^3 in volume) in 2009. The differences in SSA in the same land-use type between 1982 and 2009 were analyzed in SPSS 11.5a by independent-sample *t*-tests.

Data for 1982–2009 also included water consumption from the reservoir, water consumption from pumped groundwater, groundwater table depth, groundwater mineralization, area of exploited land, area of farmland, precipitation and evaporation. In addition, land exploitation data since 1960 from the Land Resource Administration of Fubei was used. The correlations between different factors and soil salinization were analyzed with SPSS 11.5a by bivariate correlations test (Pearson's correlation coefficients with their significance levels).

3. Results and discussion

3.1. Process of land exploitation and its environmental impact

Policy proposed by the Central Chinese Government to develop western China in the early 1960s caused much land to be reclaimed in inland river basins of northwest China. People were dispatched from all over rural China to farmland there. In the Fubei region, the chief land-use is irrigated agriculture, and overexploitation of land resources was astonishing after 1960 (Fig. 3): the area of irrigated

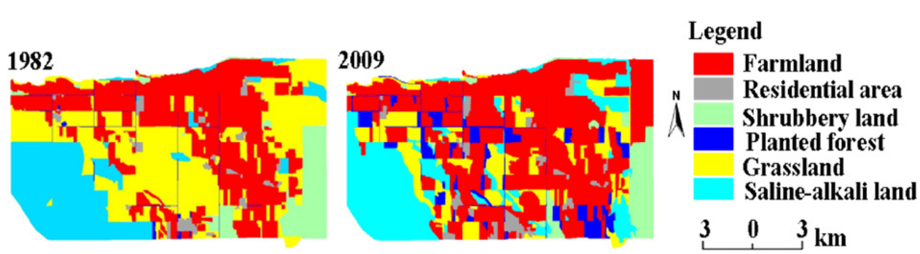

Fig. 2. Distribution of land-use type in the study area both in 1982 and in 2009.

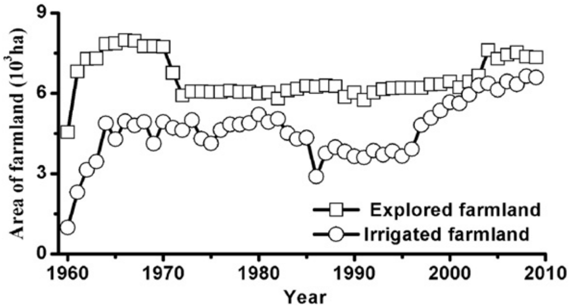

Fig. 3. The process of land exploitation from 1960 to 2009 in the study area. Explored farmland represents all land that was cultivated, including that abandoned later; irrigated farmland indicates farmland that was cultivated at that time.

farmland was < 1000 ha in the 1960s, but > 7000 ha in 2009. The area of irrigated farmland in the region increased by > 6000 ha during 1960–2009. Excessive land reclamation was obvious before 1970, with the area increasing > 3000 ha during 1960–1970. The area of land exploitation has been in a stable and slowly increasing trend since 1970, with the area of irrigated farmland in a fluctuating and increasing trend, indicating moderate land exploitation and frequent farmland abandonment. Hence, Fubei provides a natural laboratory to examine the changes in land use and its influence on soil salinization in a desert–oasis ecotone. In recent years, this region has been subject to severe soil salinization due to inappropriate land use and water resource management (Wang et al., 2007a).

3.2. Change of land-use from 1982 to 2009

Farmland dominates the Fubei region, and land-use change was mainly characterized by increasing multiplicity and fragmentation during 1982–2009 (Table 1). There has been a considerable change in the Fubei region during this 28-year-period, and approximately 38% of the total area experienced transitional changes (Table 1). The area of grassland reduced from 5272 ha in 1982 to 2581 ha in 2009. The area of saline-alkali land increased by 10.5% in 2009 compared to 1982; farmland increased by 26.6% during the same period. Furthermore, the area of planted forest increased 6.5 times in 2009 compared to 1982; however, the change in area was small, only 941 ha, which was clearly the result of land integrated management since 2000 (Wang et al., 2007b). Human activities have become a dominant factor in shaping the agricultural land-use pattern (Fu et al., 2006). Land exploitation as a socioeconomic process is the major driver of land-use change (Viglizzo et al., 2012). Land exploitation in the desert–oasis ecotone changed the distribution and composition of the Fubei regional land-use types during 1982–2009. The area of land-use change dramatically altered from one land-use type to another (Table 1). The two most prominent conversions were from grassland to farmland and from grassland to saline-alkali land (1887 and 721 ha, respectively). Conversion from grassland to planted forest and from shrub land to saline-alkali land were also notable (555 and 164 ha, respectively). These remarkable transformations were responsible for reduced areas of grassland and increased areas of farmland and saline-alkali land, indicating a faster process of oasis development than that of desertification (Lu et al., 2003).

3.3. Spatial statistical analysis of soil salt content in 1982 and 2009

The range in soil salinity values for the sample sites was 0.4–69 g/kg in 1982 (Table 2). Average values in 2009 were higher than in 1982 (12 and 8.5 g/kg, respectively). Asymmetry and kurtosis values are also provided in Table 2. Values of coefficients of variation (CV) in both 1982 and 2009 were > 40%, indicating that salt content was moderately variable in the surface soil of the study area (Sylla et al., 1995). The Kolmogorov–Smirnov (K–S) test of soil salinity variables in 1982 and 2009 suggested a normal data distribution ($p<0.05$), indicating that the data and size of the sample was appropriate for geostatistical analysis of a semivariogram (Li and Reynolds, 1995).

In the present study, a semivariogram was employed to analyze the structure and spatial and temporal variability of soil salt content (Table 3). The optimal theoretical models of soil salt content in 1982 and 2009 were exponential models. Values of R^2 were > 0.5; the Residual Sum of Square (RSS) was small (0.006 in 1982, 0.007 in 2009) and the *F*-test for R^2 was significant ($p<0.01$). These parameters indicated that the exponential models in 1982 and 2009 well reflected the spatial structural characteristics of the point soil measurements. The ratio of nugget and sill ($C0/C0+C$) was < 0.25, suggesting strong spatial autocorrelation (Cambardella et al., 1994) and indicating that spatial dependence of soil salt content was mainly due to structural factors. The spatial variability of soil salt content might be directly or indirectly due to natural and human action on soil processes in the region, e.g. irrigation, precipitation, evaporation and the rising groundwater table (Wang et al., 2008a). This inference was also consistent with the conclusion from statistics of the variabil-

Table 1
Land-use change matrix, 1982–2009 (ha).

1982	2009						
	Farmland	Residential area	Shrub land	Planted forest	Grassland	Saline-alkali land	1982 Total
Farmland	5213	55	32	349	106	43	5798
Residential area	40	292	0	14	21	8	375
Shrub land	18	86	860	75	110	164	1313
Planted forest	9	19	17	91	9	0	145
Grassland	1887	114	64	555	1931	721	5272
Saline-alkali land	172	26	18	2	404	2356	2978
2009 Total	7339	592	991	1086	2581	3292	15881
Change area	1541	217	−322	941	−2691	314	–
Percent (%)	26.6	57.9	−24.5	649	−51	10.5	–

Table 2
Statistics of soil salt content and K–S test in 1982 and 2009.

Time	Sample	Mean	SD	CV (%)	Minimum	Maximum	Skewness	Kurtosis	K–S
1982	86	8.45	4.06	48.05	0.41	60.18	1.35	1.57	2.21
2009	147	12.15	7.03	57.86	0.73	63.74	1.49	1.88	2.58

Table 3
The parameters and prediction errors of models (when fitted to an exponential semivariogram) for soil salt content in 1982 and 2009.

Time	Model	Co	Sill	Co/sill	Rang (km)	R^2	RSS	F-value	ME	RMSE
1982	Exponential	0.0234	0.1706	0.1424	1.26	0.511	0.006	7.68**	0.0011	0.959
2009	Exponential	0.0327	0.2497	0.1309	1.39	0.648	0.007	28.02**	0.0012	0.977

Note: **Means *F*-test significance at $p<0.01$.

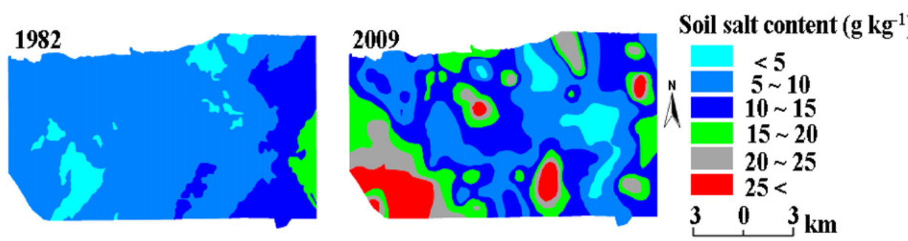

Fig. 4. Distribution map of soil salt content in the study area both in 1982 and in 2009.

ity of soil salt content that reduced grassland and increased farmland and saline-alkali land were significantly related to land-use change (Table 1).

Geostatistical analysis provides an estimation of uncertainties at unsampled locations and is a very valuable method of presenting distributions of soil salt content degree. In this study, cokriging was utilized to estimate spatial distributions of soil salinity and sampling strategies. For the spatial prediction of soil salt content variables in 1982 and 2009, the range of MEs of predictions was around 0.001 (i.e. approaching 0), and for RMSE was 0.96–0.98, indicating that the spatial prediction maps of soil salt content obtained by kriging interpolation were reliable (Chang et al., 1998). The prediction map of soil salt content in 1982 and 2009 (Fig. 4) illustrates that the distribution patterns of soil salt content in the study area changed greatly during the last 28 years, and clearly shows that soil salt content in 1982 was lower than in 2009.

3.4. Soil salinity and land-use types in 1982 and 2009

The statistical analysis of areas of different soil salt contents (Table 4) showed that, during 1982–2009, the area of soil salinization significantly increased as did the degree of soil salinity. For example, in 1982, >80% of the land (about 12,660 ha) had <10 g/kg in salt content; however in 2009, >64% of the land had >10 g/kg in salt content, and >16.4% (approximately 2606 ha) had >20 g/kg. In addition, the area of soil salt content within 5–10 g/kg decreased by 6608 ha (about 42%) in 2009 compared to 1982. Clearly, soil salinization was an astonishingly significant process in this region during 1982–2009, suggesting that the process of land exploitation changed the pattern of the land-use type at the same time as increasing soil salinization. Comparing distribution of soil salt content with land use for the same period showed that the expanding area of soil salinity and a higher degree of saline land were significant in both artificial and natural landscapes during 1982–2009. In 1982, the land-use types with higher soil salinity were only grassland and shrub land with soil salt content >15 g/kg, and accounted for only 3.2% of the Fubei region (Table 4). However, in 2009, the area of soil salinity >15 g/kg was scattered in each land-use type—in total 5172 ha or 56% of the natural landscapes fell into this category (Table 4). In addition, areas with soil salt content of >20 g/kg also occurred in different land-use types in 2009, which did not occur in 1982.

Table 4
Distribution of soil salt content within land-use types in 1982 and 2009 (ha).

Landscape type	Time	Soil salt content (g/kg)						Total
		<5	5–10	10–15	15–20	20–25	>25	
Farmland	1982	377	4846	575	0	0	0	5798
	2009	726	3189	2392	642	215	175	7339
Residential area	1982	0	315	60	0	0	0	375
	2009	43	316	134	96	3	0	592
Shrub land	1982	13	178	624	498	0	0	1313
	2009	10	54	285	497	71	74	991
Planted forest	1982	9	0	136	0	0	0	145
	2009	18	540	350	111	40	27	1086
Grassland	1982	58	4009	1191	14	0	0	5272
	2009	104	682	1023	551	139	82	2581
Saline-alkali land	1982	789	2066	123	0	0	0	2978
	2009	11	25	807	669	985	795	3292
Total	1982	1246	11,414	2709	512	0	0	15,881
	2009	912	4806	4991	2566	1453	1153	15,881

3.5. SSA from 1982 to 2009

The amount of SSA in 0–20 cm depth increased by about 21.6×10^{10} g from 1982 to 2009 in the sum of all land-use types (Table 5). The mean value of SSA in 2009 was clearly higher than in 1982, and SSA per unit area increased by about 60%, indicating an accelerating degree of soil salinization in the study area. Furthermore, the mean SSA in the same land-use types significantly differed between 1982 and 2009 ($p<0.05$). This was especially so in farmland, grassland and saline-alkali land (Table 5) where the mean value of SSA increase was $>9\times10^6$ g ha^{-1}, indicating that SSA was greater and the degree of soil salinization was serious in the surface soil of these land-use types. The conversion of natural land-use type to agricultural land had a large impact on SSA changes in time and space.

Table 5
SSA in 0–20-cm depth for different land-use types.

Land-use type	Mean ($\times10^6$ g ha^{-1})		Sum ($\times10^{10}$ g)	
	1982	2009	1982	2009
Farmland	25.19 ± 4.71	34.38 ± 4.19*	14.61 ± 2.73	25.23 ± 3.51
Residential area	22.75 ± 3.96	27.86 ± 3.93	0.85 ± 0.15	1.65 ± 0.23
Shrub land	28.34 ± 3.00	34.30 ± 3.16	3.72 ± 0.39	3.40 ± 0.31
Planted forest	38.29 ± 3.83	40.89 ± 5.33	0.56 ± 0.07	4.44 ± 0.58
Grassland	23.4 ± 3.92	34.78 ± 3.99*	12.34 ± 2.07	8.98 ± 1.03
Saline-alkali land	13.99 ± 3.08	43.08 ± 3.62*	4.17 ± 0.92	14.18 ± 1.19
Total	22.82 ± 3.98	36.45 ± 4.32*	36.24 ± 6.33	57.88 ± 6.86

Note: *Significant difference for mean SSA in same land-use type ($p<0.05$).

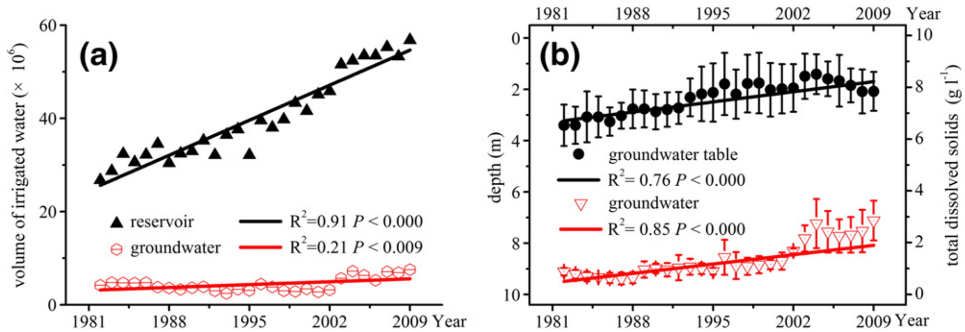

Fig. 5. Changes in water sources for irrigation and groundwater changes from 1982–2009 in the studied area. (a) The quantity of water source for irrigation, modified from Wang et al. (2008a); and (b) the groundwater table and total dissolved solids in groundwater.

3.6. Water consumption increased the farmland area and soil salinization

Water resources in the Fubei region mainly come from two sources: pumped groundwater and a reservoir via a canal into the region. With the expansion of irrigated land by 2633 ha (including area of planted forest) from 1982 to 2009, water consumption increased from 2.93×10^7 m^3 in 1982 to 6.43×10^7 m^3 in 2009 (more than doubled). There was a significant increasing trend for use of irrigated water from the reservoir in the studied periods (Fig. 5a, $p<0.000$), with >90% of the increased use of irrigation water from the reservoir. Although irrigation by pumped groundwater also increased, correlation was very weak with a lower R^2 value (Fig. 5a, $R^2=0.21$). The changes in consumption of water from the reservoir indicated that surface water via canal into the region was the main source of irrigation water. Due to the dramatic increase in the use of irrigation water from the reservoir, the level of the groundwater table and groundwater mineralization in the region increased significantly during 1982–2009 (Fig. 5b, $p<0.000$). Increased level of groundwater tables and groundwater salinization are the main factors driving soil salinization (Bennett et al., 2009; Han et al., 2011; Qadir et al., 2000).

Correlations between soil salinization and water use, as well as meteorological factors, were analyzed for the Fubei region (Table 6, $n=28$). During 1982–2009, the annual volume of pumped groundwater was not significantly correlated with either increased area of farmland or annual volume of irrigated water, suggesting that consumption of pumped groundwater was not a major contributor to irrigation of farmland. Water consumption from the reservoir was positively ($p<0.01$) correlated with the increased area of farmland and salinization of the groundwater; and negatively correlated with the rise in the groundwater table ($p<0.01$) as well as annual evaporation ($p<0.05$). This indicated that the increased farmland area resulted from more water input from the reservoir and this directly caused the rising in the groundwater table. Higher evaporation was also a driver for increased areas of saline-alkali land and increased soil salt contents (Tables 1 and 4). Therefore, much more effort is needed to decrease soil salt content by adjusting land-use structure as an effective strategy, especially to limit and properly manage irrigated farmland in the study area.

4. Concluding remarks

Overexploitation of land resources was astonishing in the Fubei region after 1960, with >40% increase in the area of irrigated farmland. The region experienced substantial and increasing rates of land-use change during 1982–2009. There have been persistent changes both spatially and temporally, resulting in 38% of the total area experiencing transitional changes among land-use types. The general trend in the study area implies a loss of grassland and shrub land cover and an increase in cultivated areas and saline-alkali land. Land exploitation raised the groundwater table through irrigation leaching, and then caused soil salinization. The expanding area of soil salinity and higher degree of saline land were significant in all land-use types of the study area, with a 16.4% increase in the area with soil salt content >20 g/kg during 1982–2009. In addition, the SSA increased by 21.6×10^{10} g over all land-use types during 1982–2009 in the study area. SSA per unit area increased 60%, with the highest accumulation in farmland, grassland and saline-alkali land. The expansion of irrigated farmland required more water to be transported from the reservoir into the study area, and this directly caused the rising groundwater table, and then higher evaporation led to soil salinization. The present trend may lead to more soil salinization if no appropriate measures are taken to stop this process. Continued land-use change, coupled with soil salinization, has greatly affected people's livelihoods and put the crop production system under increasing threat. The results of the present study not only elucidate the changes in land use and the impact on soil salinization in the Fubei region, but represent a lesson to be learned for integrated land management in similar ecotones in arid zones.

Table 6
Correlations from Pearson's test of different factors and soil salinization ($n=28$).

	X1 (ha)	*X2* (m^3)	*X3* (m^3)	*X4* (m^3)	*X5* (mm)	*X6* (mm)	*X7* (g L^{-1})	*X8* (m)
X1 (ha)	1							
X2 (m^3)	0.87**	1						
X3 (m^3)	0.86**	0.99**	1					
X4 (m^3)	0.30	0.35	0.27	1				
X5 (mm)	−0.20	−0.17	−0.17	0.37	1			
X6 (mm)	0.42*	0.47*	0.43*	0.22	−0.30	1		
X7 (g L^{-1})	0.86**	0.92**	0.93**	−0.05	−0.22	0.41*	1	
X8 (m)	−0.70**	−0.86**	−0.88**	−0.03	0.08	−0.14	−0.83**	1

Note: *, **Significant at $p<0.05$ and $p<0.01$, respectively. *X1* is the area of irrigated farmland, *X2* is the annual volume of the irrigated water, *X3* is the annual volume of the irrigated water from the reservoir, *X4* is the annual volume of irrigated water from pumped groundwater, *X5* is the annual precipitation, *X6* is the annual pan evaporation, *X7* is the annual average of groundwater TDS (total dissolved solids), and *X8* is the annual average groundwater table.

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