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RESEARCH ARTICLE

Linking wind erosion to ecosystem services in drylands: a landscape ecological approach

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Abstract
Context Wind erosion is a widespread environmental problem in the world's arid landscapes, which threatens the sustainability of ecosystem services in these regions.
Objectives We investigated how wind erosion and key ecosystem services changed concurrently and what major biophysical and socioeconomic factors were responsible for these changes in a dryland area of China.
Methods Based on remote sensing data, field measurements, and modeling, we quantified the spatiotemporal patterns of both wind erosion and four key ecosystem services (soil conservation, crop production, meat production, and carbon storage) in the Mu Us Sandy Land in northern China during 2000–2013. Linear regression was used to explore possible relationships between wind erosion and ecosystem services.
Results From 2000 to 2013, wind erosion decreased by as much as 60% and the four ecosystem services all increased substantially. These trends were attributable to vegetation recovery due mainly to government-aided ecological restoration projects and, to a lesser degree, slightly increasing precipitation and decreasing wind speed during the second half of the study period. The maximum soil loss dropped an order of magnitude when vegetation cover increased from 10% to 30%, halved again when vegetation increased

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from 30 to 40%, and showed little change when vegetation increased beyond 60%.
Conclusions Our study indicates that vegetation cover has nonlinear and threshold effects on wind erosion through constraining the maximum soil loss, which further affects dryland ecosystem services. These findings have important implications for ecological restoration and ecosystem management in dryland landscapes in China and beyond.

Keywords Wind erosion · Ecosystem services · Drylands · Constraint effect · Mu Us Sandy Land, Inner Mongolia

Introduction

Ecosystem services (i.e., benefits that people derive from nature), as a concept, has been widely used in academia as well as resource management and policy-making to bridge ecology and economics and to connect ecosystems and human well-being (Dominati et al. 2010; Gómez-Baggethun et al. 2010; Butler et al. 2013; Wu 2013; Byrd et al. 2015; Zhang et al. 2015; Bagstad et al. 2017; Kukkala and Moilanen 2017). Four types of ecosystem services are widely recognized: supporting services (i.e., ecosystem processes or functions), provisioning services (e.g., food, water, shelter), regulating services (e.g., air and water purification, climate modification, carbon sequestration), and cultural services (e.g., recreation, traditional heritage) (MEA 2005; Wu 2013). Soils are the foundations of terrestrial ecosystems and, as a form of natural capital, provide a multitude of important goods and services to society (Blum 2005; Dominati et al. 2010). Changes in soil properties affect ecosystem processes, such as nutrient cycling and water fluxes, which in turn influence ecosystem services (Fu et al. 2013; Adhikari and Hartemink 2016; Guerra et al. 2016). Thus, it is imperative to link soils and ecosystem services so as to better understand their interrelations and improve soil protection measures to ensure the production and delivery of these services for human needs (Adhikari and Hartemink 2016; Calzolari et al. 2016).

Drylands, home to 38% of global human population and covering 41% of the world's land area, are faced with myriad sustainability challenges (Reynolds et al. 2007; MEA 2005). Wind erosion is one of the most common forms of soil degradation in drylands worldwide (Lancaster and Baas 1998; Shi et al. 2004; Buschiazzo and Zobeck 2008; Shao 2008; Harper et al. 2010; Hoffmann et al. 2011; Leenders et al. 2011; Borrelli et al. 2016). A large number of studies have been carried out to investigate how vegetation and landscape patterns affect wind erosion and how soil erosion influences ecosystem functions (Zobeck et al. 2000; Lal 2003; Shi et al. 2004; Yan et al. 2005, 2013; Buschiazzo et al. 2007; Li et al. 2007; Harper et al. 2010; Vanacker et al. 2014). Now soil erosion is widely recognized as one of the greatest threats to sustainable ecosystem services through both physical and biological pathways (Larney et al. 1998; de Rouw and Rajot 2004; Wang et al. 2006; Li et al. 2009; Harper et al. 2010). Wind-blown sands and dusts, removed from disrupted soil aggregates or even soil horizons by wind, can directly damage vegetation, resulting in the decline or loss of plant production and vegetation's ability to protect soils from erosion (Zuazo and Pleguezuelo 2008; Li et al. 2009; Dominati et al. 2010).

As wind erosion decreases the stocks of nutrients and organic matter, consequently reducing soil fertility and soil functions, it in turn diminishes ecosystem services of the affected areas. Although it seems obvious that soil erosion decreases ecosystem services in general, it is far from clear whether the effects of soil erosion on ecosystem services are linear/gradual or nonlinear/with thresholds. It is also unclear if soil erosion affects the different types of ecosystem services in similar ways. While the negative impacts of wind erosion on soil ecosystem functions have been widely documented in many drylands around the world, how soil erosion and its impacts on ecosystem processes further affect dryland ecosystem services remains poorly understood (Munson et al. 2011). Some of the recent wind erosion-related studies examined ecosystem services impacts, but focused primarily on the supporting services which are essentially ecosystem processes or functions (MEA 2005; Wu 2013). Much research is needed to improve our understanding of how soil erosion affects other types of ecosystem services. In addition, most of the previous studies on the relationship between soil erosion and ecosystem services focused on single scales, but a multiscale perspective is needed for a comprehensive understanding of ecosystem services dynamics in time and space (Wu 2013).

Therefore, the main objectives of this study were to quantify the spatiotemporal patterns of wind erosion and key ecosystem services and to examine their relationship in a dryland area in northern China using a landscape ecological approach. Specifically, we addressed the following research questions: (1) How did wind erosion intensity and ecosystem services (crop and meat production, carbon storage, and soil conservation) change in time and space from 2000 to 2013? (2) How did wind erosion affect ecosystem services in time and space? (3) How did major biophysical factors affect wind erosion, and what lessons can be learnt to reduce soil erosion and enhance ecosystem services in this dryland region?

Materials and methods

Study area

Our study area was the Mu Us Sandy Land region (Fig. 1), located in the southern part of the Ordos Plateau which is mostly within Inner Mongolia where extensive drylands are found (Fang et al. 2015; Wu et al. 2015). The entire study region covers a total area of 86,000 km^2, including the Mu Us Sandy Land itself (about 42,200 km^2) and neighboring counties of Inner Mongolia, Shaanxi, and Ningxia. The Mu Us Sandy Land was formed as the result of a long-term soil sandification driven mainly by climate change and geophysical processes, but since the 1800 s human activities have become an increasingly important factor to its expansion (Wu et al. 2015). The area has a typical continental semi-arid climate, and much of

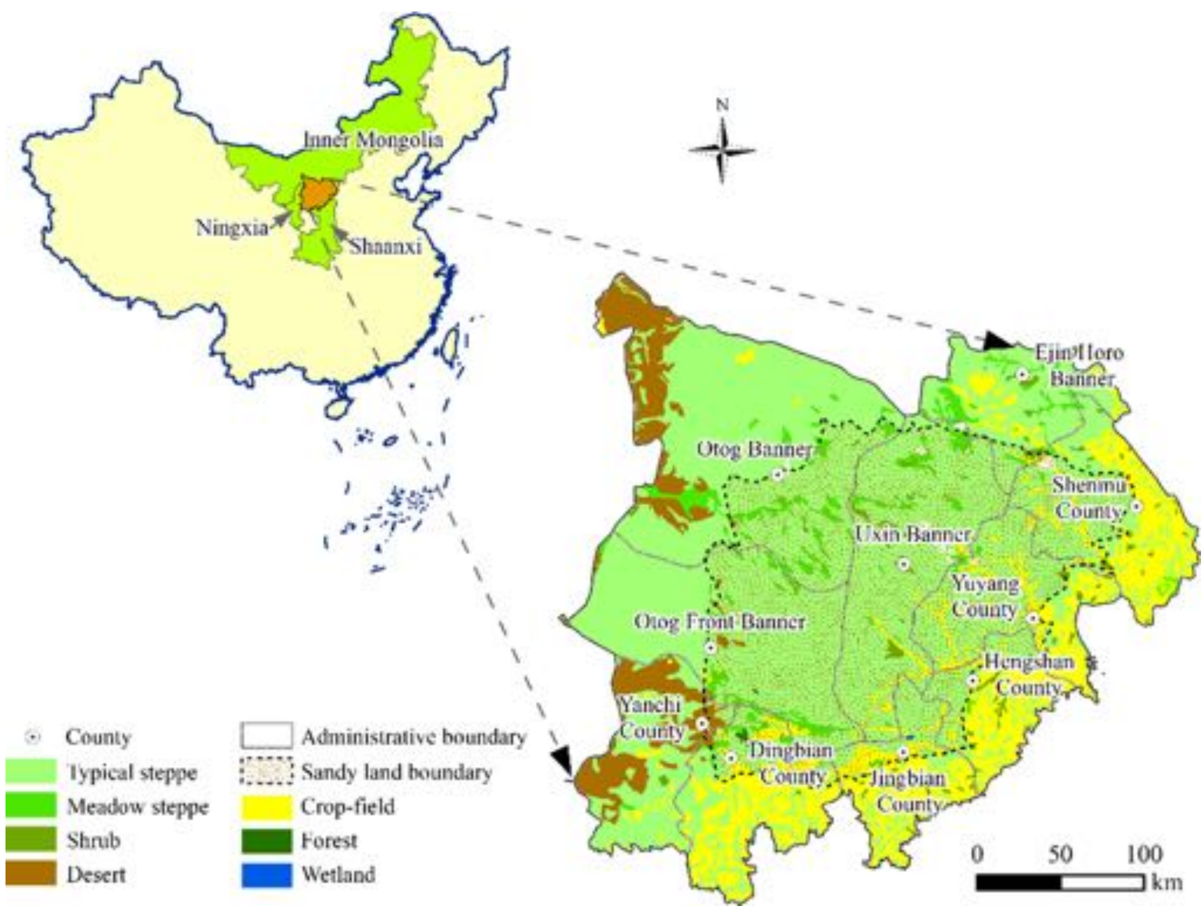


Fig. 1 A locational map of the study area located in the southern part of the Ordos Plateau, China, consisting of the Mu Us Sandy Land and its vicinity

the region is classified as part of the typical steppe zone (Wu and Loucks 1992). The annual precipitation ranges from 250 to 400 mm and the annual mean temperature is about 6–8.5 °C (Yan et al. 2015). Potential evaporation is 2220 mm annually, and hence the aridity index (a measure of the dryness of the climate) indicates an arid environment (Karnieli et al. 2014). The landscape features of the Mu Us Sandy Land are characterized by shifting dunes, semi-fixed dunes, fixed dunes, and low ridged-land, with multiple vegetation types of widely distributed grasslands and deserts as well as localized forests and crop-fields (Fig. 1).

Traditional land uses in the Mu Us Sandy Land have been livestock grazing and farming, a typical combination in the agro-pastoral transitional zone of northern China (Wu et al. 2015). Sandy grasslands cover more than 80% of the region with *Artemisia Ordosica* as a dominant species (Yan et al. 2015). Grasslands in the northwestern part of the Mu Us Sandy Land are mainly used for grazing, but some grasslands in the eastern and southern parts have been converted into farmland. Most of the farmlands are distributed along the southern and eastern borders (Fig. 1).

The Mu Us Sandy Land is one of the most seriously degraded areas in China (Wu et al. 2015; Zhou et al. 2015). The dry and windy weather during spring often cause severe wind erosion and frequent dust storm events (Yue et al. 2015). Because of the ecological fragility of these drylands and the increasing pressures from regional population growth and economic development, restoring the soil capital and maintaining ecosystem services in the Mu Us Sandy Land have become increasingly urgent for the sustainable development of the region (Fang et al. 2015; Wu et al. 2015; Yue et al. 2015).

Data

Meteorological data (i.e. daily wind speed, rainfall, temperature and total solar radiation) from weather stations in and around the study area were obtained from the China Meteorological Data Sharing Service System (http://cdc.nmic.cn/home.do). Snow cover data were downloaded from the Environmental and Ecological Science Data Center for West China, National Natural Science Foundation of China (EESDC, NNSFC) (http://westdac.westgis.ac.cn/). Data of soil types and soil texture were obtained from the Harmonized World Soil Database provided by the EESDC (Fischer et al. 2008). The three most dominant soil types in the Mu Us Sandy Land region are the haplic arenosols, cambic arenosols, and calcaric arenosols.

The vegetation pattern of the study area was obtained from the digitized vegetation map of China at the cartographic scale of 1: 1,000,000 (The Editorial Committee of Vegetation Map of China of CAS 2007). We classified the vegetation into seven types: typical steppe, meadow steppe, shrub, desert, crop-field, forest, and wetland (Fig. 1). Normalized Difference Vegetation Index (NDVI) data from 2000 to 2013 were derived from the SPOT/VGT S10 dataset (http://www.vito-eodata.be). We produced an annual NDVI time series with a spatial resolution of 1 km by combining 10-day NDVI values using the maximum value composite method.

Data on crop yields and meat production of each county in the region between 2000 and 2013 were obtained from the statistical yearbooks of Inner Mongolia, Ningxia and Shaanxi compiled. The crops includes rice, wheat, corn, sorghum, grains, and tuber crops. Meat includes pork, mutton, and beef. The land use/cover dataset in 2013 at the scale of 1:100,000 was obtained from the China's Land- Use/cover Datasets (Liu et al. 2014b). The administrative boundary data at the scale of 1:1,000,000 were from the National Administration of Surveying, Mapping and Geoinformation. All the data were interpolated or resampled to 90 × 90 m to avoid scale-mismatching problems (Wu et al. 2006).

Conceptual framework for linking wind erosion and ecosystem services

To guide our study, we developed a framework that identifies key measures and indicators of wind erosion, ecosystem services, and environmental drivers, and indicates their relationships and statistical methods used to quantify them (Fig. 2). The framework reflects a landscape ecological perspective that emphasizes the pattern-process-driver dynamics of the wind erosion-ecosystem services relationship on multiple scales.

Climatic factors (e.g., wind velocity and direction, temperature, and precipitation) and vegetation cover are widely recognized key biophysical factors affecting soil erosion by wind. In the wind erosion modeling literature, climatic factors are lumped into a single "weather factor", which is mainly a function of wind

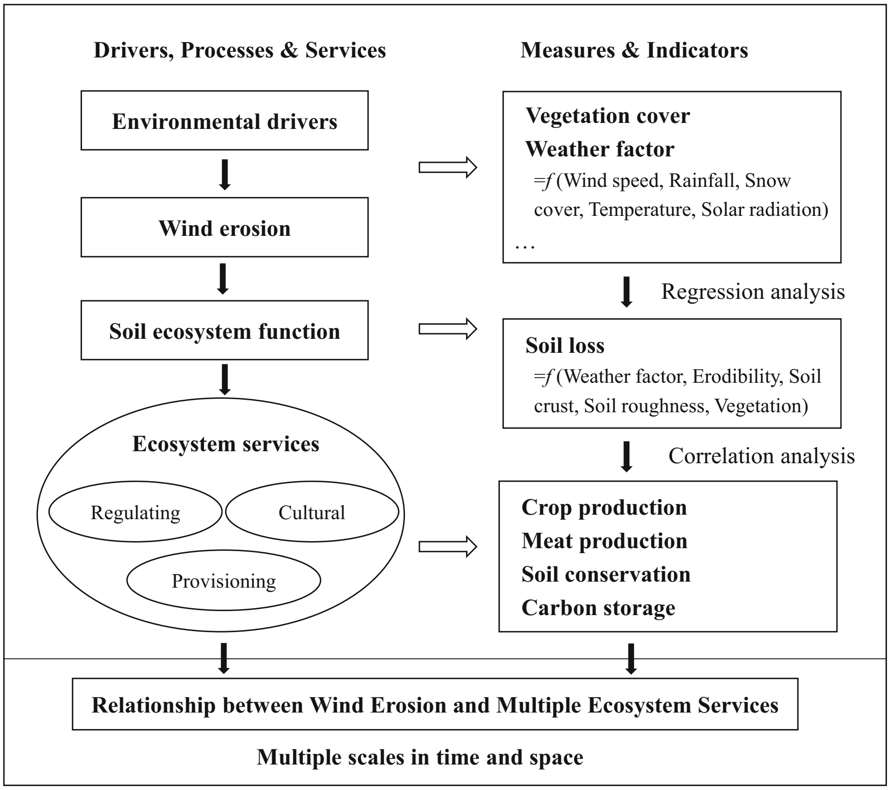


Fig. 2 A framework for linking wind erosion and ecosystem services in dryland landscapes, showing the relationship among environmental drivers, wind erosion, and ecosystem services (left column) as well as the corresponding measures/ indicators and statistical methods used for the analysis (right column)

speed, solar radiation, rainfall, and snow cover (Fryrear et al. 2000). Thus, the weather factor represents the overall driving force of the wind erosion process. The intensity of wind erosion is often measured by the amount of soil loss per unit of time—a commonly used indicator for soil degradation. While all three types of ecosystem services—provisioning, regulating, and cultural—are important, this study focused on two provisioning services (crop and meat production) and two regulating services (soil conservation and carbon storage). We did not consider cultural services here because of the lack of available data. The four ecosystem services considered here are fundamentally important, and together should be adequately indicative of the overall environmental quality of the region.

Quantifying wind erosion and ecosystem services

Quantifying soil loss by wind erosion

To measure the intensity of wind erosion, we estimated soil loss using the revised wind erosion equation (RWEQ) which requires data on climate, soil, and management (Fryrear et al. 2000; Gong et al. 2014b; Borrelli et al. 2016):

$$Q_x = 2xQ_{max}\exp(-(x/s)^2)/s^2 \tag{1}$$

$$Q_{max} = 109.8(WF \times EF \times SCF \times K' \times COG) \tag{2}$$

$$s = 150.71(WF \times EF \times SCF \times K' \times COG)^{-0.3711} \tag{3}$$

where Q_x is the amount of soil transported to a distance × downwind from the upwind boundary; x is the distance from the upwind edge of the field; Q_{max} is the maximum transport capacity; s is critical field length; WF is the weather factor; EF is the erodible fraction of surface soil; SCF is the soil crust factor; K' is the soil roughness factor (mainly determined by land cover types); and COG is the combined vegetation factor (estimated from NDVI) (Gong et al. 2014b).

The estimation of all the factors in the revised wind erosion equation can be found in Fryrear et al. (2000) and Gong et al. (2014b). Following the methods in Fryrear et al. (2000) and Gong et al. (2014b), we calculated WF using data on daily wind speed, rainfall, temperature, total solar radiation, and snow depth; determined EF and SCF according to the soil contents of sand, silt, clay, organic matter, and calcium carbonate; and estimated K' based on different land cover types. Moreover, we obtained vegetation cover for half-month periods based on NDVI using the dimidiate pixel model (Gutman and Ignatov 1998), and calculated COG for determining the effect of withered and growing vegetation (Gong et al. 2014b). Soil loss was computed for 15-day time periods (Fryrear et al. 2000), and then summed all them up to get the annual total soil loss from 2000 to 2013.

The RWEQ model was previously applied in several studies from northern China (e.g., Guo et al. 2013; Gong et al. 2014b). Gong et al. (2014a, 2014b) used the model to simulate the semi-arid grassland region in northern China, including the Hunshandac Sandy Land of Inner Mongolia, and found that their simulated results were adequately accurate as compared to the field measurements. We used the RWEQ model in the same way, and derived all the parameters from our study region. Our simulated annual soil loss in Yuyang County in 2005 compared reasonably well with the estimate by Yue et al. (2015) who used a different approach that combined remote sensing data with sand transport modeling. Specifically, the estimated soil loss of Yuyang County in 2005 was 1859.41 t/(km^2 year) by our model, which was 10.35% higher than that by Yue et al. (2015).

Quantifying ecosystem services

We used crop production and meat production from statistical yearbooks to represent provisioning services. The county-level crop production and meat production during 2000–2013 were mapped to show their spatiotemporal patterns.

The soil conservation rate (SCR) during wind erosion was defined as (Gong et al. 2014b):

$$SCR = (Q_p - Q_x)/Q_p \tag{4}$$

where Q_x is defined the same way as in the revised wind erosion equation, and Q_p is the amount of potential soil erosion under bare soil conditions, which can be calculated using Eqs. (1–3).

Carbon storage includes the aboveground biomass, underground biomass, and soil carbon storage. It is usually calculated as the product of carbon density and the vegetated area. We used a conversion factor of 0.45 to convert the biomass to carbon content (Fang et al. 2007). The aboveground biomass was calculated using annual maximum NDVI:

$$B = a \times NDVI^b \tag{5}$$

where B is the biomass; a is 179.71 and 8.5582 for grasslands and croplands, respectively; and b is 1.6228 and 2.4201 for grasslands and croplands, respectively (Fang et al. 2007). The R^2 of the regression for grasslands and croplands was 0.71 and 0.62, respectively (Fang et al. 2007).

The aboveground biomass model was validated with field data and used for assessing carbon stocks in China's terrestrial vegetation (Fang et al. 2007). The underground biomass carbon density and soil carbon density were then converted from the aboveground biomass carbon density based on the ratio given in previous studies (Olson et al. 1983; Piao et al. 2004).

Statistical analysis

We used linear regression to analyze the temporal changes in average wind erosion intensity and ecosystem services at the county level, with the slope of the regression line representing the annual change rate. The determination coefficient of the regression was used to indicate the strength of the relationship between wind erosion or ecosystem services which both changed in time and space. The analysis was performed at two spatial scales: the whole study region and the county level.

We examined the major drivers of wind erosion using constraint line analysis (Thomson et al. 1996; Guo et al. 1998) and multiple linear regression. In bivariate scattergrams, data points sometimes show clouds bounded by an informative edge, implying that the independent variable may act as a limiting factor constraining the response of the dependent variable ("constraint effect"). In this case, constraint line analysis has been suggested in lieu of traditional correlation and regression methods (Thomson et al. 1996; Guo et al. 1998; Wang et al. 2016). This was the case for the scattergrams of wind erosion versus vegetation cover in our study. Thus, we quantified the impacts of vegetation cover on wind erosion using the constraint line method as in Wang et al. (2016).

We further conducted a multiple linear regression analysis to quantify the relative contributions of different biophysical factors to soil loss, with vegetation cover and the weather factor as independent variables. The standardized regression coefficient (SRC) was calculated using the standard deviations of variables to represent changes in the outcome associated with a unit change in the predictor variable (Ma et al. 2016). The larger the absolute value of the SRC, the more important that independent variable.

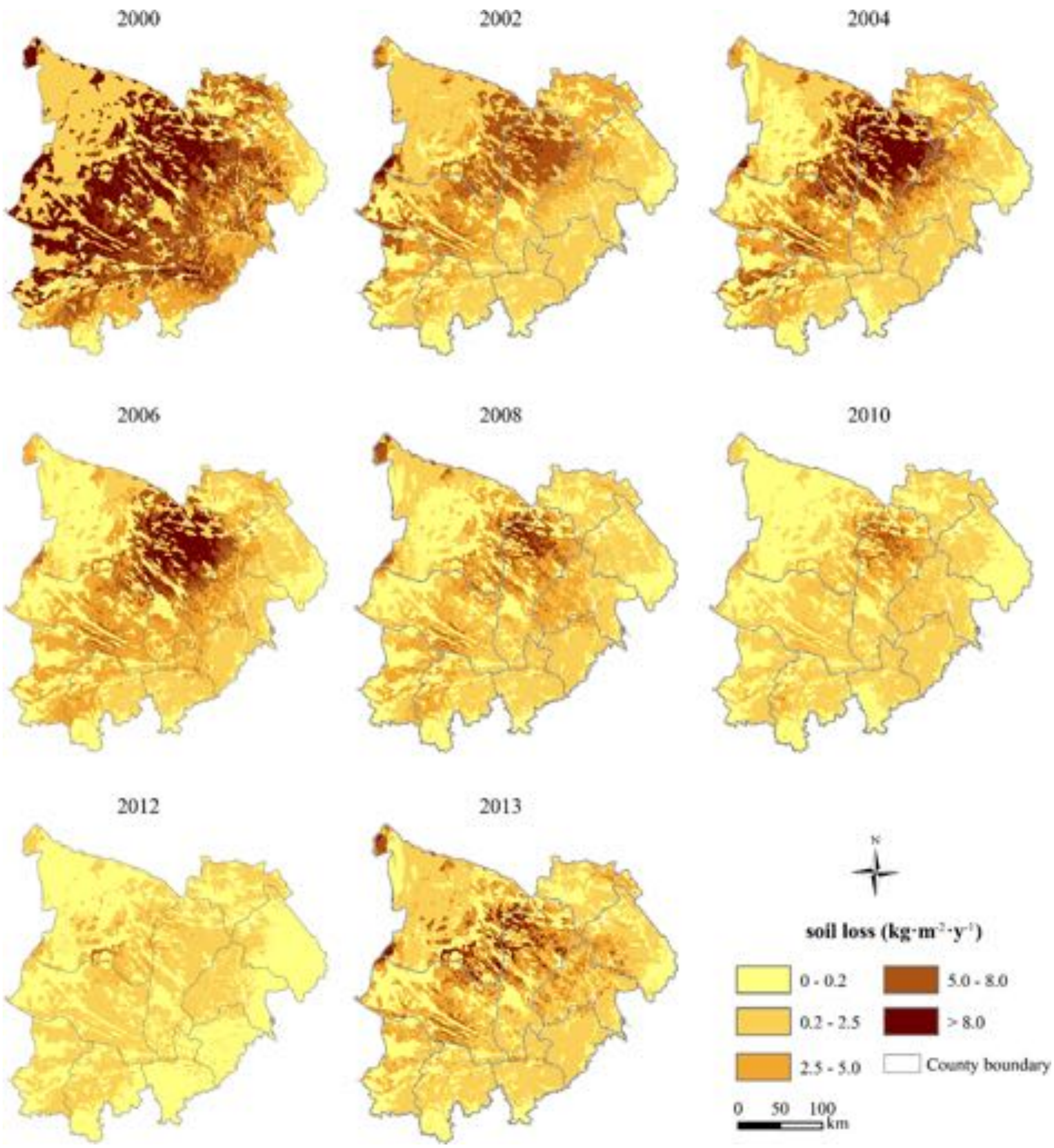


Fig. 3 Spatiotemporal patterns of soil loss in the Mu Us Sandy Land region from 2000 to 2013

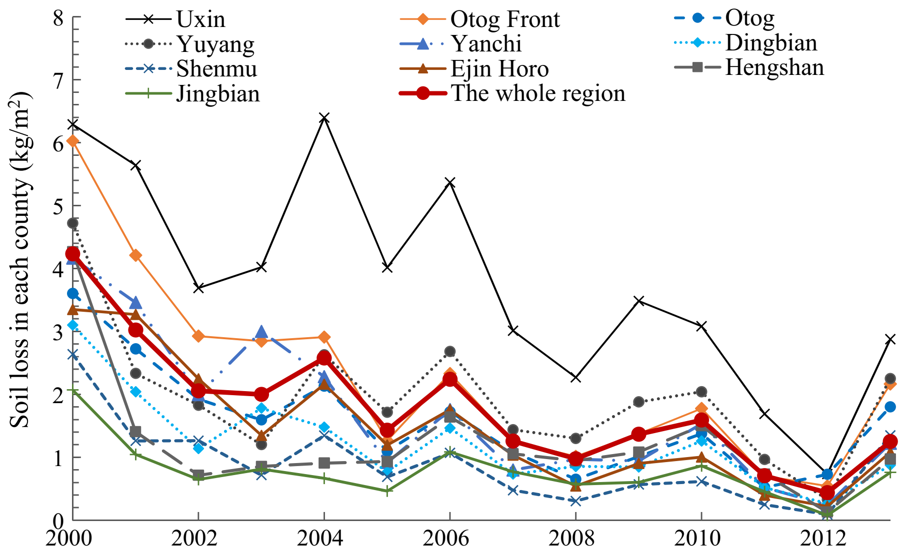


Fig. 4 Temporal changes of soil loss in the Mu Us Sandy Land region from 2000 to 2013

All of the statistical analyses were performed with SPSS for Windows.

Results

Spatiotemporal patterns of wind erosion

Wind erosion varied considerably from 2000 to 2013 in space (Fig. 3) and time (Fig. 4). The average soil loss density of the whole region in 2013 was 1698 t km^{-2} - $year^{-1}$. According to the standards of soil erosion classes published by the Ministry of Water Resources of the People's Republic of China, about 70% of the region had a "weak" or "moderate" level of wind erosion, with soil loss of < 2500 t km^{-2} $year^{-1}$. About 8% of the region experienced "highly intense" or "most intense" wind erosion, with soil loss of > 5000 t km^{-2} $year^{-1}$ (Figs. 3, 4). These places were located mainly in Uxin Banner (2184.4 km^2, 2.54%) and Otog Banner (2829.4 km^2, 3.29%) (Fig. 3).

The total soil loss in the Mu Us Sandy Land region fluctuated during the study period, but exhibited a statistically significant decreasing trend over the 14 years, dropping from 372.26×10^9 kg in 2000 to 108.10×10^9 kg in 2013, with the minimum of 36.67×10^9 kg in 2012 (Fig. 4). The annual decrease of soil loss during the 14 years was 17.20×10^9 kg a^{-1}, and the total decrease accounted for about 60% of the soil loss in 2000.

The relationship between biophysical drivers and wind erosion

The scatter plots of soil loss against vegetation cover showed point clouds with relatively obvious informative boundaries (Fig. 5). Our constraint line analysis quantified the constraint (or limiting) effect of vegetation cover on soil loss. For both the whole region and each county, the maximum soil loss decreased exponentially with increasing vegetation cover (Fig. 5). When vegetation cover increased from 10 to 30%, the maximum soil loss decreased sharply from more than a thousand to lower than 150 t km^{-2}; when vegetation cover increased from 30 to 40%, soil loss decreased by another half; and when vegetation cover reached about 60%, wind erosion was essentially undetectable (Fig. 5).

The further multiple linear regression analysis showed that the SRC value of weather factor (SRC_w) was much larger than that of vegetation factor (SRC_v) (Table 1). For the Mu Us Sandy Land region, the SRC_w value (0.914) was 6.72 times SRC_v (0.136). We further explored the relative contribution of weather factor under different values of vegetation cover. With vegetation cover increasing from 20 to 60%, the ratio

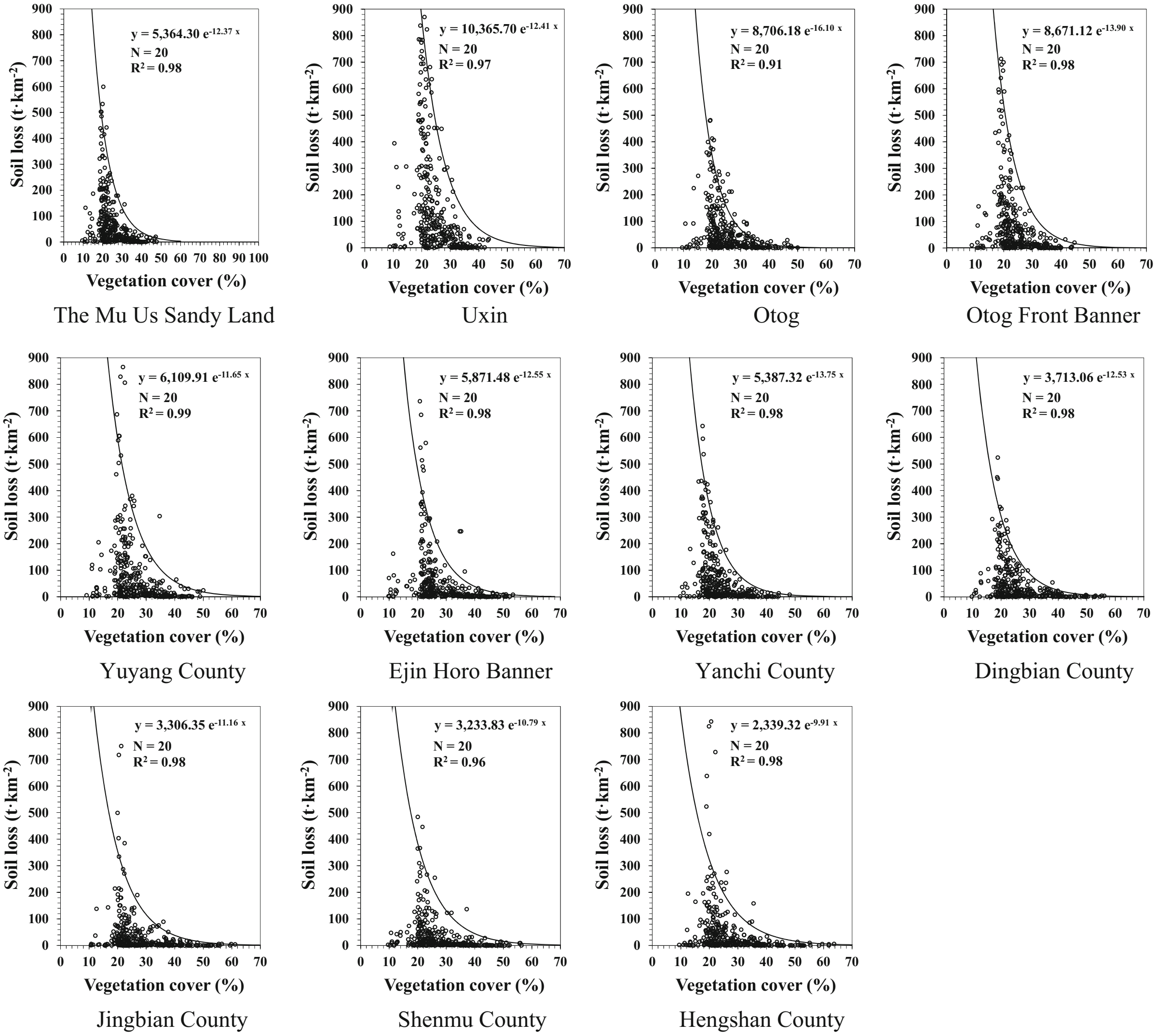


Fig. 5 The relationship between soil loss and vegetation cover (soil loss represented by 15-day values)

Table 1 Standardized regression coefficients (SRC) between soil loss and its driving factors. SRC_w and SRC_v denote the standardized regression coefficients of the weather factor and vegetation cover, respectively

County/Banner	SRC_v	SRC_w	R^2	Times (SRC_w/SRC_v)
Otog	0.110	0.934	0.94	8.49
Otog Front	0.117	0.934	0.94	7.98
Jingbian	0.120	0.895	0.85	7.46
Hengshan	0.120	0.894	0.85	7.45
Ejin Horo	0.121	0.899	0.88	7.43
Yanchi	0.135	0.910	0.92	6.74
Yuyang	0.148	0.903	0.89	6.10
Shenmu	0.147	0.869	0.83	5.91
Uxin	0.171	0.858	0.84	5.02
Dingbian	1.018	0.693	0.21	0.68
The Mu Us Sandy Land region	0.136	0.914	0.93	6.72

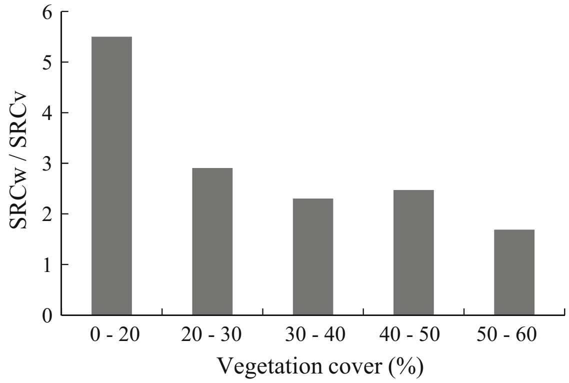


Fig. 6 The relative contributions of the weather factor and vegetation cover to soil loss due to wind erosion. SRC_w and SRC_v denote the standardized regression coefficients of the weather factor and vegetation cover, respectively

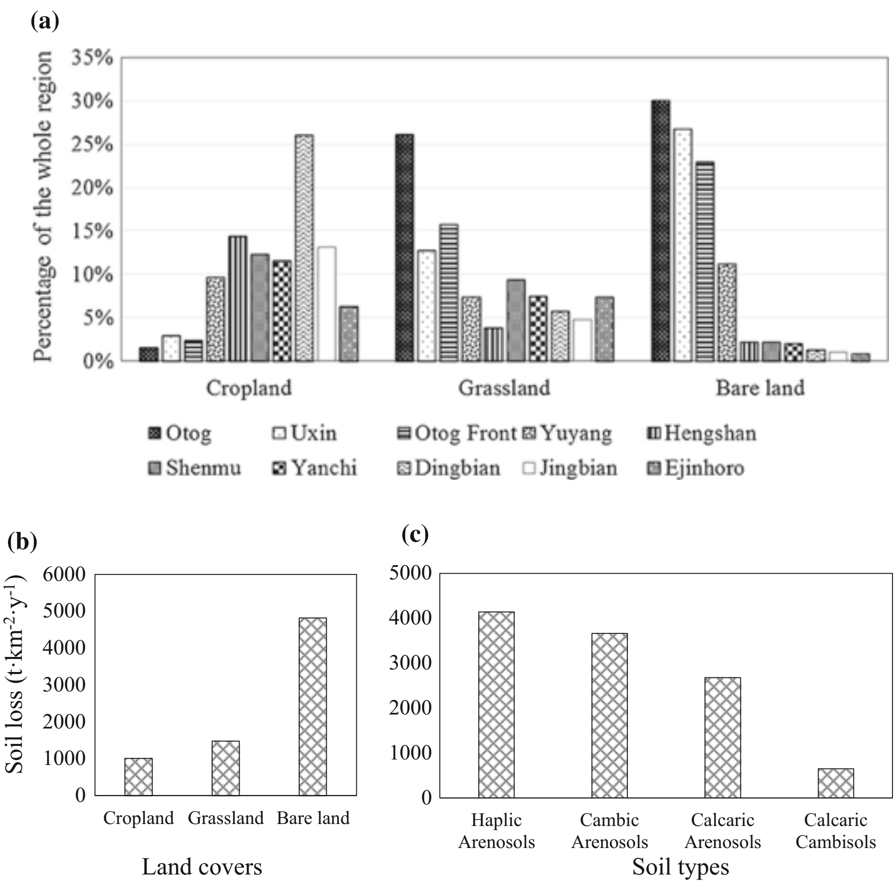
of SRC_w to SRC_v showed a decreasing trend (Fig. 6). SRC_v was not statistically significant when vegetation cover was over 60%.

By overlaying the maps of soil loss with maps of land cover and soil types, we further examined how land cover and soil types were related to wind erosion (Fig. 7). In 2013, the region was composed mainly of grasslands (73.0%), croplands (14.5%) and barren lands (9.7%). For the Mu Us Sandy Land region, soil loss intensity due to wind erosion was greatest from barren lands, and least from croplands, with grasslands in the middle (Fig. 7). Soil loss intensity from the soil of haplic arenosols was greater than that from cambic arenosols, calcaric arenosols and calcaric cambisols (Fig. 7).

Spatiotemporal patterns of ecosystem services

All the four ecosystem services showed a similar increasing trend, but differed in spatial pattern from 2000 to 2013 (Figs. 8, 9). The total crop production of the Mu Us Sandy Land region increased from 7.14×10^8 kg in 2000 to 15.74×10^8 kg in 2013.

Fig. 7 The distribution of land cover types in different counties (**a**) and soil loss associated with different land cover types (**b**) and soil types (**c**) in 2013



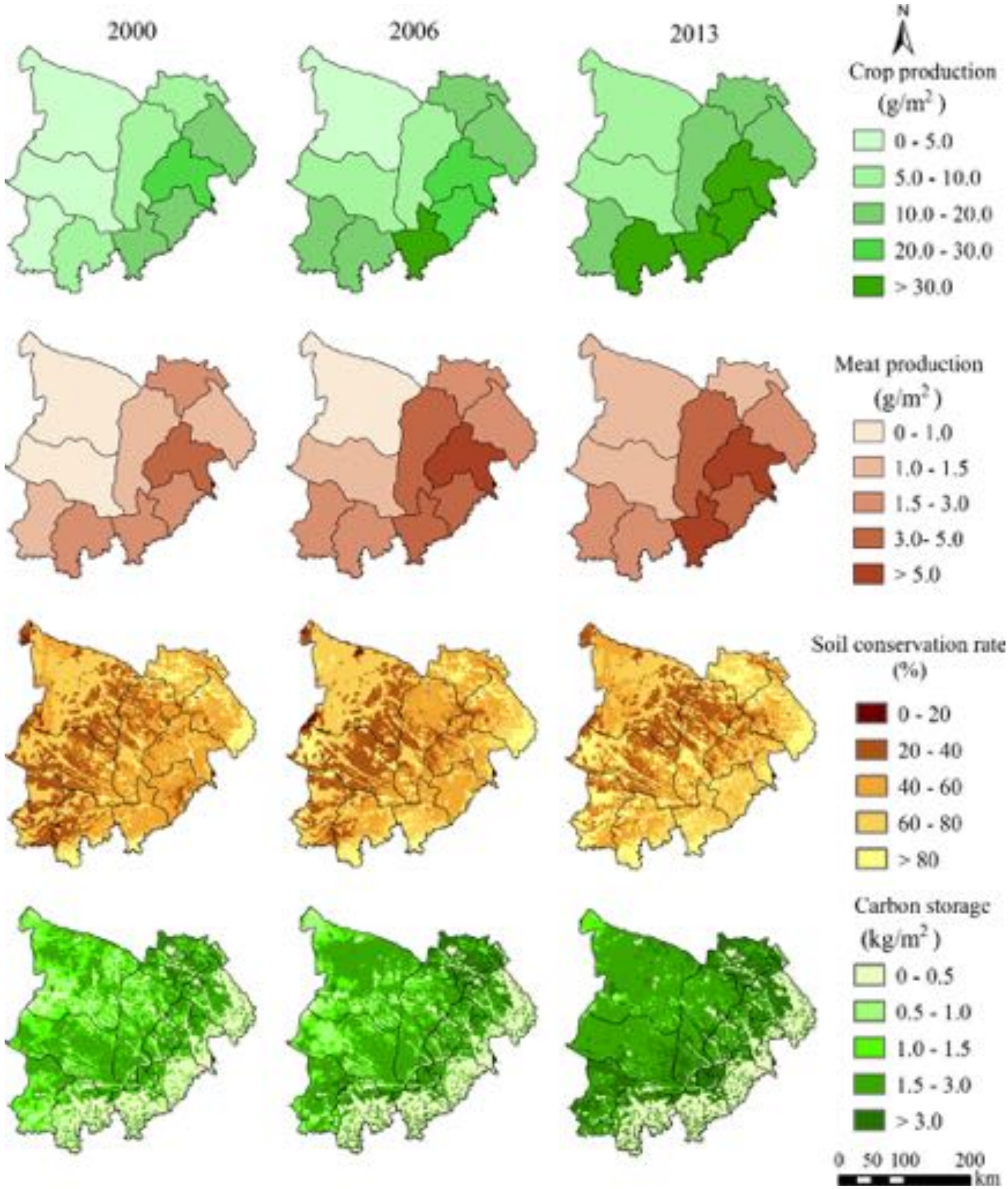


Fig. 8 Spatiotemporal patterns of ecosystem services in the Mu Us Sandy Land region from 2000 to 2013. Crop and meat productions are shown by the administrative unit of county, whereas the soil conservation rate and carbon storage are shown at the resolution of 90 m regardless of the administrative boundaries

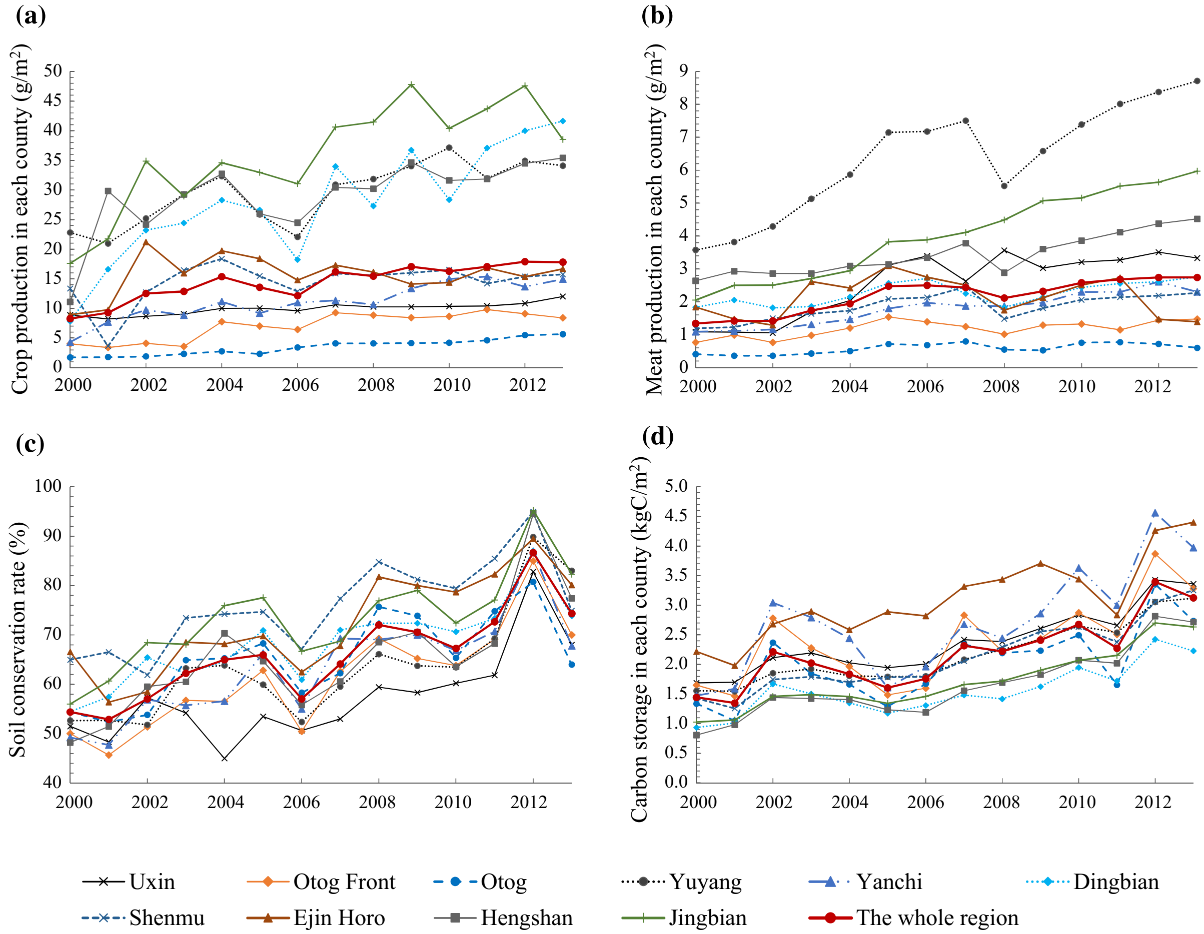


Fig. 9 Temporal changes of four key ecosystem services in the Mu Us Sandy Land region, China from 2000 to 2013

Both the crop production and its increase rate were greater in southern and eastern counties than the rest of the region (Fig. 8). About 50% of the crop production came from three counties: Dingbian, Jingbian, and Yuyang. The annual change of the three counties was 1.99×10^3, 1.80×10^3 and 0.99×10^3 kg km^{-2}, respectively, higher than the regional average (0.66×10^3 kg km^{-2}).

The total meat production of the Mu Us Sandy Land region increased from 116×10^6 kg in 2000 to 246×10^6 kg in 2013, with an annual increase rate of 9.88×10^6 kg. Uxin Banner and Yuyang County were the major contributors, accounting for 40–50% of the total meat production of the region (Figs. 8, 9).

The soil conservation capacity of the region, measured as SCR, significantly improved from 2000 to 2013, with an annual increase rate of 1.66 percentage points. Some western counties had lower soil conservation rates, but they experienced the most substantial improvements during the 14 years (Figs. 8, 9). For example, Otog Front Banner increased its soil conservation rate by 2.43 percentage points each year. Yanchi County experienced an improving trend of soil conservation with the highest determination coefficient (0.72).

The total carbon storage of the whole region increased from 125 to 270 TgC, with an annual increase of 10.41 TgC, indicating a progressive process of carbon sequestration (Figs. 8, 9). The rate of change in carbon storage was generally greater in the western counties than the rest of the region. Yanchi County in the southwestern corner of the region

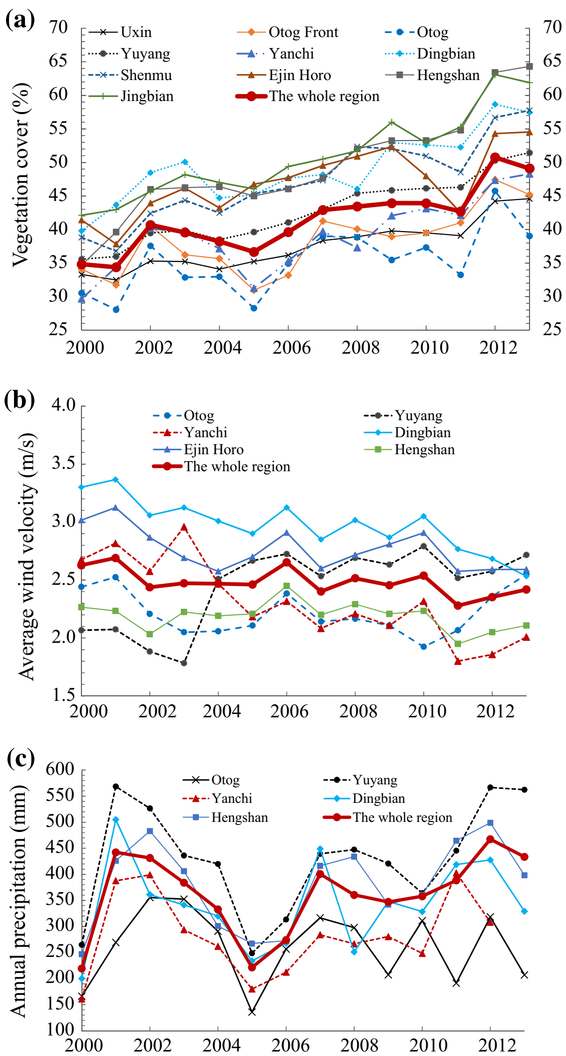


Fig. 10 Temporal changes of major biophysical factors influencing wind erosion in each county and the entire Mu Us Sandy Land region from 2000 to 2013: **a** vegetation cover, **b** average wind speed, and **c** annual precipitation based on the local National Weather Stations

experienced the greatest change, with an annual increase rate of 167 gC/m^2. The carbon storage of Uxin Banner and Yuyang County in the middle part of the sandy land showed a growing trend with higher determination coefficients.

Discussion

How did wind erosion and ecosystem services change in space and time in the Mu Us Sandy Land region?

General trends in wind erosion and ecosystem services

Our results show that the soil loss in the Mu Us Sandy Land region generally decreased during the 14 years from 2000 to 2013, with generally increasing vegetation cover, variable but slightly decreasing wind speed, and fluctuating annual precipitation (Fig. 10). This trend corresponds to recent studies reporting on the improvement of vegetation in the region during the recent decades (Zhang et al. 2014; Zhao et al. 2015; John et al. 2016). Both favorable changes in precipitation and governmental policies for land restoration in the recent decades helped the recovery and expansion of vegetation in the Mu Us Sandy Land region, thus resulting in a generally decreasing trend in soil loss and an increasing trend in the four ecosystem services (Figs. 4, 9).

Effects of wind erosion on key ecosystem services

A number of studies have shown that wind erosion affects soil texture, soil fertility, soil biodiversity, and soil ecosystem function (Li et al. 2007, 2009; Yan et al. 2013; Adhikari and Hartemink 2016). Undoubtedly, these changes further affect the kinds and amounts of ecosystem services in these areas. In our analysis, one of the four ecosystem services, soil conservation, was estimated using the wind erosion model, and thus correlation analysis between wind erosion and soil conservation in this case would not be appropriate. However, by definition wind erosion and soil conservation are conversely related to each other, and thus the decreasing trend in wind erosion in our study region inevitably suggests an increasing trend of soil conservation. In the following, our discussion is focused on crop production, meat production, and carbon storage, which were estimated independently of the wind erosion model (see the Methods section for detail).

Crop production and meat production are the two primary provisioning services in the Mu Us Sandy Land region, and our results show that both of them increased with decreasing wind erosion (Fig. 11).

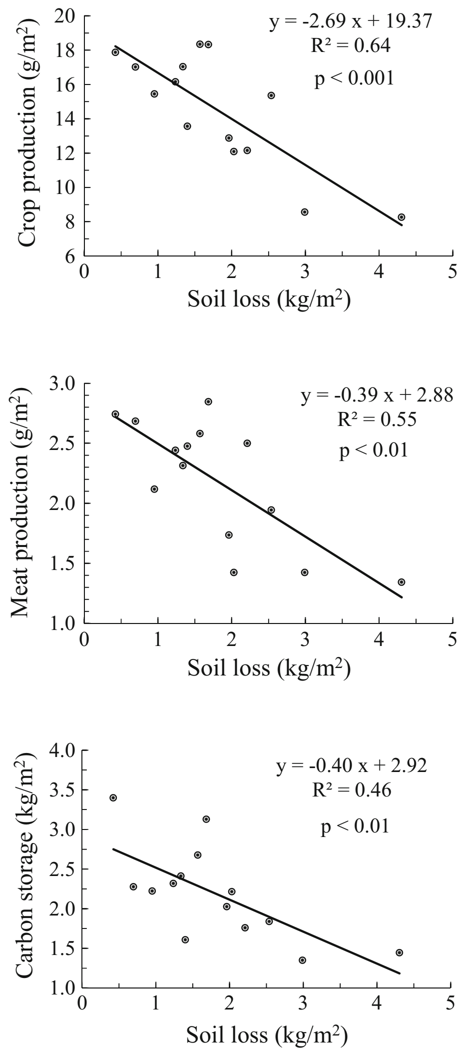


Fig. 11 Regressions between soil loss and three ecosystem services (crop production, meat production and carbon storage) in the Mu Us Sandy Land region

From 2000 to 2013, the crop production of the region doubled with reducing wind erosion. Dingbian, Hengshan, and Jingbian were three counties with extensive farmlands, and their crop productions increased 4.3, 2.1, and 1.7 times during the 14 years, respectively (Fig. 9a). The cropland area in these counties only increased by less than 1.5% during the same period. Conservation tillage measures and shelter forests helped to reduce wind erosion and thus improve soil physical properties (e.g. soil temperature, soil bulk density, soil water content), consequently increasing crop yield (Lei et al. 2008). Although the fertilizer use doubled according to available statistical data, wind erosion reduction may also have been an important indirect driving factor. Meat production was also negatively correlated with soil loss in several counties and for the entire study region (Fig. 11). The grazing-ban and grassland restoration projects were implemented since the 1990s in the region. Raising sheep in feeding lots was adopted to help balance the needs for both the herders' livelihoods and grassland sustainability. Thus, the increasing stocking rate did not lead to more severe grassland degradation and greater erosion. In the same time, the local meat production actually increased due mainly to the increase in forage production and the improved grassland conditions. In other words, wind erosion was linked to the meat production through grassland ecosystem functions (particularly grass productivity).

Carbon storage in most counties increased with declining soil loss, resulting in a strong negative correlation between the two variables at the regional level (Fig. 11). For example, the vegetation in Uxin Banner was dominated by the temperate typical steppe whose ecosystem functions are tightly coupled with precipitation in the semiarid climate. The area had many barren sand dunes, some of which were restored with planted vegetation during 2000–2013 (Zhang et al. 2014). The soil here was sandy and liable to soil erosion in that strong wind can easily blow nutrients away, consequently reducing vegetation cover and carbon storage in the biomass and soil. The improvement in soil conservation and the decrease in soil loss were especially pronounced in the "Demonstration Areas of Ecological Construction" designated by the Chinese government, suggesting that recovered vegetation was a key factor.

How did key biophysical factors affect wind erosion?

Effects of vegetation cover on wind erosion

The equations of wind erosion (Eqs. 1–3) clearly show that several factors influence wind erosion. These factors include vegetation cover (the key resistance), weather factor (the key driving force), land cover types (influencing soil surface roughness that affects the movement and impacts of soil particles as well as the development of wind profiles), and soil physical properties (determining soil erodibility or resistance to erosion). Below we discuss how these factors affected wind erosion and ecosystem services in the Mu Us Sandy Land region, starting with the effects of vegetation.

Vegetation has long been recognized as a key factor in protecting soils from wind erosion through increasing surface roughness and absorbing the downward momentum of the ambient air stream (Wasson and Nanninga 1986; Li et al. 2005). Previous studies reported that the total amount of soil loss decreased exponentially with increasing vegetation cover (Lancaster and Baas 1998; Yan et al. 2011), suggesting a strong correlation between the two variables. However, our results indicate that vegetation cover has a nonlinear "constraint effect" on soil loss, meaning that vegetation acts as a limiting factor to wind erosion that is influenced simultaneously by multiple factors. Thus, vegetation cover alone cannot predict the actual amount of soil loss without considering other key factors. What can be determined in this case is a "boundary line" or "constraint line" that approximates the "maximum soil loss" with changing vegetation cover. In our study, we found that an exponential decay function yielded the constraint line of best fit (Fig. 5). Note that the exponential decay function does not cover the entire range of vegetation cover from 0 to 100%; but rather it represents the constraint line of best fit for the range of vegetation cover within which soil loss declines from the peak to null. In a similar vein, Munson et al. (2011) reported that aeolian sediment flux declined exponentially with increasing perennial vegetation canopy cover on the Colorado Plateau. The peak soil loss at about 20% of vegetation cover may be attributable to the "funnel effect" (winds tend to speed up as they squeeze through gaps of sparse plants), while the low soil loss below 20% of vegetation cover may be a consequence of both the weakening funnel effect and the lack of existing topsoil in barren or sparsely vegetated areas. A better understanding of this phenomenon requires field-based observational and experimental studies in the future.

Our results also suggest that vegetation cover had a lower and an upper threshold for controlling wind erosion in the Mu Us Sand Land region, and the maximum soil loss declined precipitously with increasing vegetation cover between these two threshold values (Fig. 5). Specifically, the lower threshold of vegetation cover was about 20%, below which vegetation had little effect on soil erosion. This suggests that plant cover lower than 20% did little to reduce wind velocity at the soil surface. The upper threshold was about 60%, beyond which soil erosion was essentially stopped, implying that the effect of vegetation on reducing wind erosion basically reached the maximum when plant cover was 60% or above.

Our findings are in general agreement with the predictions of percolation theory and empirical observations with animal movements and fire spread in landscape ecological studies (Gardner et al. 1987; Turner et al. 2001). In particular, our estimated upper threshold of vegetation cover (60%) matches extremely well with the values determined by experiments: 61.7% from a field experiment (Yue et al. (2015) and 60% from a wind-tunnel experiment (Dong et al. 1996).

Effects of the weather factor on wind erosion

The dense point cloud under the upper edge in Fig. 5 was indicative of the important impacts of factors other than vegetation cover on wind erosion. How did the driving force of wind erosion, i.e., the weather factor (WF), affect the soil loss in the Mu Us Sandy Land region as compared to vegetation cover? Wind speed is the main factor in sediment transport, and the interaction of strong wind with dry, loose soil surface can cause serious erosion (Shao 2008). Precipitation and temperature have important effects on soil erodibility (McKenna Neuman 2003). Thus, weather factor is mainly a function of wind speed, rainfall, and temperature. Our study showed that the weather factor had a greater overall contribution to soil loss than vegetation cover at both the regional and county scales, as indicated by their different absolute values of the standardized regression coefficient ($SRC_w > -SRC_v$). However, the relative contribution of the weather factor to wind erosion decreased with increasing vegetation cover although its absolute contribution was always greater than that of vegetation (Fig. 6). In other words, the influence of the weather factor on soil loss is modulated by vegetation, and this modulation is strong when vegetation cover is between 20 and 60%.

Effects of land cover and soil types on wind erosion

Land cover types and soil properties also contributed to the spatial pattern of soil loss density in the study region. As compared to climatic factors and vegetation cover, soil factors and land cover types usually affect soil erosion on longer time scales because their dynamics are slower. Different soil types that vary in texture, mineralogy, chemistry and organic matter content influence soil particle sizes and weight, and their ability to retain moisture and form bounds, all of which were important for determining soil erodibility (Webb and Strong, 2011). Soil physical and chemical characteristics vary greatly between different land use/covers such as grasslands and croplands (Rezaei et al. 2016). For the Mu Us Sandy Land region, soil loss due to wind erosion was greatest from barren lands (Fig. 7b). Barren lands were covered mostly by the soil types of Cambic Arenosols and Haplic Arenosols, which are more vulnerable to wind erosion (Fig. 7c). About 80% of barren lands in the study region were concentrated in three counties (Otog, Uxin, and Otog Front Banner) (Fig. 7a). This was an important reason why these counties experienced more severe wind erosion.

What lessons can be learnt to help reduce wind erosion and thus improve ecosystem services?

In order to reduce soil erosion and improve ecosystem services, the Chinese government has implemented a number of wind erosion mitigation projects since the late 1950s, including the Three-North Shelterbelt Project (1979–2050), the Grain-to-Green Project (1999–2010), and the Beijing and Tianjin Sandstorm Source (BTSS) Control Project (2001–present). These projects have been met with success in many areas in northern China (e.g., Gao et al. 2012; Liu et al. 2014a; Zhang et al. 2014; Hu et al. 2015; Wu et al. 2015), but their efficiency or cost–benefit ratio can certainly be improved. Towards that end, our study provides important lessons especially for ecological restoration on local and regional scales.

First of all, because the Mu Us Sandy Land region is characterized by arid and semiarid climates, sandy soils, and relatively sparse vegetation, large-scale cultivation should be prohibited, overgrazing by livestock should be prevented, and large-area tree-planting should be discouraged. It is clear from our study and many other studies that reducing wind erosion requires reducing the area of sandy lands without vegetation. This in turn requires re-vegetation that can take place naturally within several years in many areas of the region if human disturbances are removed (Zhang et al. 2014; Wu et al. 2015). However, the limited precipitation cannot support large areas of trees in a long-term, and tree-planting campaigns in the arid and semiarid regions of Inner Mongolia have done little to help prevent wind erosion (Wu et al. 2015).

Second, human-aided re-vegetation projects are needed to reduce wind erosion and improve ecosystem services in the region. Some projects of this kind, as mentioned above, have already been implemented in the recent decades, but more are needed, with mainly shrubs and herbaceous plants native to the region and with higher ecological and economic efficiencies. Our study indicates that there are important threshold vegetation cover values should be considered to improve the efficiency of re-vegetation efforts. If planted vegetation cover is too high, the costs can be prohibitive. If the cover is too low, the vegetation does little to stop wind erosion. As a general guide, our study suggests that vegetation cover should be at least higher than 20%, but there is no need to exceed 60% when planting vegetation to reduce wind erosion. However, the exact optimal vegetation cover for revegetation is expected to change from place to place, and it should be determined locally by considering other factors such as soil conditions, topography, plant species, and their spatial patterns.

Limitations and future directions

A few limitations of our study ought to be addressed in future studies. First, we used the RWEQ model to estimate wind erosion for the entire study region, without being able to directly validate the results because of the lack of empirical data. The most reliable data of wind erosion would come from direct field measurements or experiments, but unfortunately such data do not exist for our study region. We did derive the parameters of the model from biophysical data in the study region, and found that our estimates of wind erosion were comparable with other independent studies in the same region or under similar climatic conditions.

Overall, our estimated values of wind erosion and ecosystem services must contain considerable uncertainties, and thus the emphasis of our study was placed on comparing spatiotemporal patterns, instead of point predictions in space or time. As such, our main findings seem robust. In the future, high-resolution remote sensing data of vegetation, soil, topography, and other biophysical factors can be used to provide independent estimates of wind erosion, and help calibrate and validate wind erosion models. Most ideally, regional-scale monitoring networks for wind erosion would be able to provide more direct measurements.

Second, soil loss and soil conservation were estimated using the same wind erosion model, which would invalidate any statistical analysis because of the problem of variable interdependence or circular reasoning. Fortunately, soil loss and soil conservation should always be conversely related both theoretically and practically. Third, this study considered only four key ecosystem services in the Mu Us Sandy Land region, but also other regulating (e.g., air quality, water yield, and water purification) and cultural services (recreation and minority traditions) should be studied in the future. Finally, our statistical analysis on the effects of wind erosion on ecosystem services suggests possible the mechanisms behind the effects although correlation is not causation. Field-based, process-oriented studies are needed to verify these effects and understand the underlying mechanisms.

Conclusions

In the Mu Us Sandy Land region, wind erosion decreased and key ecosystem services (crop production, meat production, carbon storage, and soil conservation), in general, increased from 2000 to 2013. During the 14 years, wind erosion decreased by as much as 60%, while crop production, meat production, and carbon storage more than doubled their amounts, with the soil conservation rate increasing by 20%. Vegetation recovery due mainly to government-aided ecological restoration projects, as well as slightly increasing precipitation and decreasing wind speed during the second half of the study period, may all have contributed to these trends. Land cover and soil types also contributed to the changing spatial pattern of wind erosion and ecosystem services in the region. Our results suggest that wind erosion strongly affected key ecosystem services in this dryland region although the detailed mechanisms demand further studies. In contrast with previous studies, our study shows that vegetation cover affected wind erosion through constraining the maximum soil loss with multiple thresholds. These findings can help design more ecologically and economically efficient policies for reducing wind erosion and improving ecosystem services in this dryland region and beyond.

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