


RESEARCH

Open Access



Spatiotemporal changes and driving factors of ecosystem services between karst and non-karst World Heritage sites in Southwest China

Yue Chen^{1†}, Li Rong^{1†}, Kangning Xiong^{1*}, Mingjun Feng¹ and Cai Cheng²

Abstract

Understanding the spatiotemporal variation and drivers of ecosystem services is fundamental to optimal management and sustainable development of World Heritage (WH) sites. Although WH sites face multiple natural and anthropogenic threats, our understanding of their ecosystem services is still limited, especially for karst WH sites. In this study, we assessed habitat quality (HQ), carbon storage (CS), soil retention (SR), water conservation (WC), and the combined ecosystem service (CES) of karst and non-karst WH sites in Southwest China from 2000 to 2020 using the InVEST model. We also assessed trade-offs/synergies among ecosystem services using the spatial overlay method, and identified driving factors of variation in ecosystem services using geographical detector and structural equation models. The results showed that ecosystem services of the WH sites exhibited high spatiotemporal variation. In particular, there were higher values in the property zone than in the buffer zone, and an increasing trend in SR but a decreasing trend in HQ and CES over time. Compared to non-karst sites, karst WH sites had significantly lower values of HQ, CS, SR, and CES, but higher spatial heterogeneity in CS, WC, and CES. Weak trade-offs among ecosystem services dominated the WH sites, with the proportion of weak synergies increasing over time. Compared to non-karst sites, karst WH sites had a significantly lower proportion of strong synergies and a significantly higher proportion of weak synergies. The provision of ecosystem services was primarily influenced by natural factors (e.g., landscape division index and normalized difference vegetation index), followed by anthropogenic factors (e.g., distance from road and population density). Overall, these findings may have important implications for decision-making aimed at protecting the outstanding universal value, authenticity, and integrity of WH with different attributes.

Keywords Ecosystem services, InVEST model, Karst, Structural equation modeling, Trade-offs/synergies, World Heritage

[†]Yue Chen and Li Rong have contributed equally to this work.

*Correspondence:

Kangning Xiong
xiongkn@gznu.edu.cn

Full list of author information is available at the end of the article

Introduction

Accounting for about 15% of the global land area, karst is a distinctive landscape generated from the interactions between water and soluble rocks (e.g., limestone and dolomite), forming highly diverse but vulnerable environments at the surface and underground [1, 2]. Covering more than 550,000 km², the karst area in Southwest China represents the most typical and complex tropical-subtropical karst development, which is also the largest and most concentrated karst ecologically fragile area in the world [3, 4]. South China Karst is a serial World Heritage (WH) nomination submitted by the State Party China to the UNESCO World Heritage Committee [5]. Phase I was inscribed on the WH List in 2007 and comprises three internationally acclaimed areas: Shilin Karst, Libo Karst, and Wulong Karst. Phase II was inscribed in 2014 and includes Guilin Karst, Shibing Karst, Jinfo Shan Karst, and Huanjiang Karst as an extension of Libo Karst. Due to the fragility, sensitivity, and low natural regeneration capacity of karst resources and environments [6], karst WH ecosystems are more vulnerable to degradation and less likely to recover once degraded than non-karst counterparts [7–9]. This poses a serious threat to the outstanding universal value, authenticity, and integrity of South China Karst [10].

While WH sites are prioritized for ecological conservation, they face serious threats from natural factors and human pressures, such as climate change, species invasions, tourism, and land use change [11–13]. For example, although the accreditation of WH is oriented to identify, recognize, and protect hotspots of outstanding universal value, it has increasingly been utilized as a marketing tool for tourism promotion due to the benefits of the “WH” brand [14–16]. Once an area is inscribed on the WH List, it usually becomes a well-known tourist destination, with an order of magnitude more tourist visits than unlisted destinations [17]. These threats inevitably alter the structure and processes of WH ecosystems, resulting in habitat deterioration, biodiversity loss, degenerated ecosystem functions, and ultimately declined ecosystem services [18]. Therefore, information on the spatiotemporal changes and driving factors of ecosystem services is needed to capture the ecological environment of WH sites. This can help to design timely and effective management strategies that aim to reconcile conflicts between ecological conservation priorities and economic development demands in WH sites.

Ecosystem services refer to all direct and indirect benefits that humans derive from the natural environment [19]. These services are influenced by multiple natural and anthropogenic factors, such as land use, topography, climate, vegetation, soil, and human activities [20–22]. Due to the outstanding universal value of biodiversity

and the natural environment, WH sites can provide multiple regulating, supporting, and cultural services to humankind. However, different services may interact antagonistically or synergistically, resulting in trade-offs or synergies between ecosystem services [23]. Wang et al. [24] reported that carbon storage, habitat quality, aesthetic value, and recreational value of the Bogda WH site showed significant spatiotemporal variation, with higher values in central forests and high-coverage grasslands. Sui et al. [25] found that forests were crucial for maintaining habitat quality at the Jiuzhaigou WH site. They also found that ecological restoration projects effectively improved habitat quality after geohazards [25]. Wang et al. [26] showed that the ecosystem service value of the Bayanbulak WH site declined over time as a result of habitat degradation in the buffer zone. While these studies provide important information on the ecosystem services of non-karst WH sites, little is known about the spatiotemporal changes, trade-offs/synergies, and driving factors of the ecosystem services provided by karst WH sites [27]. These knowledge gaps may hinder our ability to balance the conservation and development of karst WH sites.

In this study, we used the InVEST tool to investigate spatiotemporal changes in four key ecosystem services—habitat quality (HQ), carbon storage (CS), soil retention (SR), and water conservation (WC)—of karst and non-karst WH sites in Southwest China. We also used the spatial overlay method to assess trade-offs/synergies among ecosystem services. Finally, we used geographical detector model (GDM) and structural equation modeling (SEM) to identify driving factors of variation in ecosystem services. The objectives of this study were (1) to assess the spatiotemporal variation in ecosystem services of karst and non-karst WH sites from 2000 to 2020, (2) to explore trade-offs and synergies among ecosystem services of karst and non-karst WH sites, and (3) to identify the key driving factors influencing ecosystem services. By comparing ecosystem services between karst and non-karst WH sites and revealing the key factors driving these differences, our study would advance our understanding of ecosystem services delivered by karst WH sites and provide constructive suggestions for optimal management and sustainable development of WH with different attributes.

Materials and methods

Study area

We selected two typical and representative karst WH sites (Shibing Karst and Libo-Huanjiang Karst) as the study area, with two non-karst WH sites (Fanjingshan and Chishui Danxia) as the reference (Fig. 1). Shibing Karst and Libo-Huanjiang Karst are the components of

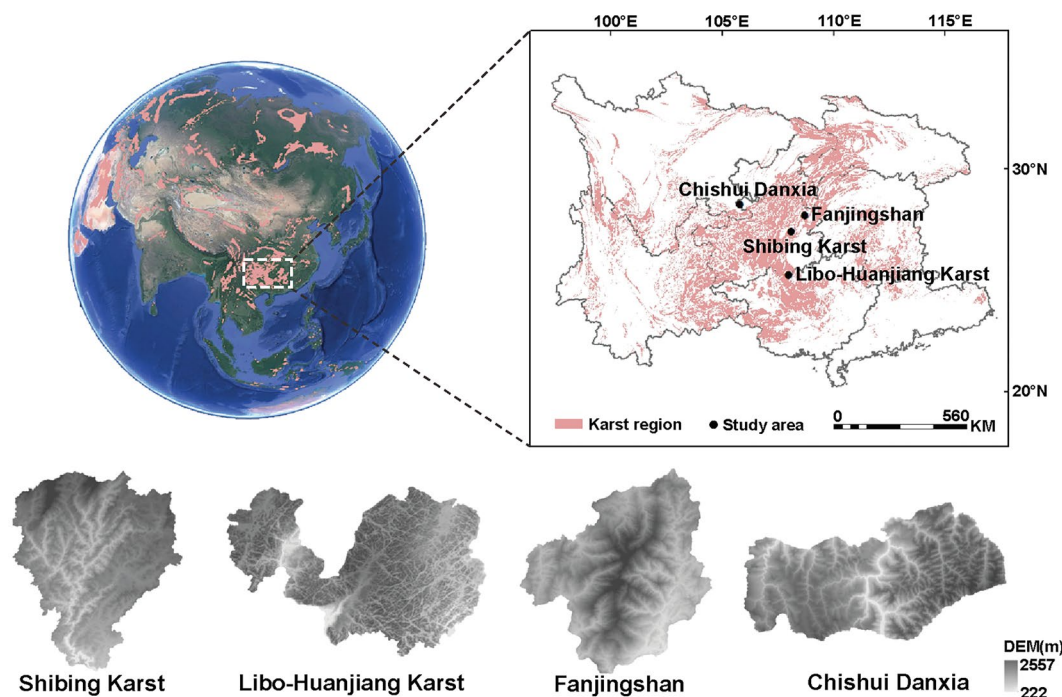


Fig. 1 Distribution of the four World Heritage sites in Southwest China

South China Karst, which is the most spectacular example of humid tropical to subtropical karst landscapes in the world. Shibing Karst provides a world reference for dolomite karst in tropical and subtropical regions [28]. Libo-Huanjiang Karst is the best example of cone karst in the world [29]. Fanjingshan contains unique biological and ecological value, which is an outstanding representative of the evolution of subtropical mountain forests [30]. Chishui Danxia preserves a typical subtropical evergreen broad-leaved forest, which is an ideal place to study the ancient and modern vegetation on sandstone [31]. Overall, these four WH sites not only include outstanding universal value in terms of natural beauty, geological features, and biodiversity, but also provide a wide range of ecosystem services to humankind.

Data sources

We collected multi-source data to assess spatiotemporal changes and driving factors of ecosystem services provided by the four WH sites (Table 1). Specifically, we evaluated ecosystem services based on land use data, biophysical parameterization, meteorological, topographic, and soil data. We explored the driving factors of ecosystem services based on climate, topography, landscape, soil, vegetation, and human activity data. Land use data were divided into six classes: cropland, forest, shrubland, grassland, water body, and impervious land. Data in non-raster formats, such as climate data, were processed using

the kriging spatial interpolation in ArcGIS 10.2 software for spatial mapping. All data were converted to a uniform projection coordinate system (WGS-1984) and were resampled to a spatial resolution of 30 m.

Ecosystem service assessment

Based on the environmental conditions and ecological functions of WH sites, we evaluated four key ecosystem services (i.e., HQ, CS, SR, and WC) using the InVEST model (version 3.12), which is one of the most widely used tools for ecosystem service assessment [21, 32, 33]. Specifically, we used the habitat quality module for HQ, the carbon storage module for CS, the sediment delivery ratio module for SR, and the water yield module for WC. The detailed parameters were provided in Tables S1–S3 [34–43] and the detailed equations for each service were as follows.

Habitat quality

HQ refers to the ability of ecosystems to provide the necessary conditions for the survival and breeding of species, which is key to the protection of biodiversity. The habitat quality module of the InVEST tool evaluates HQ using the habitat suitability, influencing distance and weight of threat sources, and the sensitivity of each habitat type to threat sources [32, 44]. HQ was calculated as:

Table 1 Information on the data used to assess ecosystem services and identify the driving factors

Data	Type	Year	Resolution	Source
Land use	Raster	2000/2010/2020	30 m	ZENEDO (https://www.doi.org/10.5281/zenodo.8176941)
DEM	Raster	–	30 m	National Ecosystem Science Data Center (http://www.nesdc.org.cn)
NDVI	Raster	2000/2010/2020	30 m	National Ecosystem Science Data Center (http://www.nesdc.org.cn)
Soil	Raster	–	90 m, 1000 m	Nation Earth System Science Data Center (https://www.geodata.cn), Harmonized World Soil Database (http://westdc.westgis.ac.cn)
Population	Raster	2000/2010/2020	100 m	WorldPop (https://www.worldpop.org/)
Nighttime light	Raster	2000/2010/2020	500 m	Nation Earth System Science Data Center (https://www.geodata.cn)
GDP	Raster	2000/2010/2020	1000 m	Resource and Environmental Science Data Platform (https://www.resdc.cn/)
Road	Vector	–	30 m	OpenStreetMap (https://download.geofabrik.de)
Meteorological data	Txt	2000/2010/2020	–	Meteorological Data Service Center, China (https://data.cma.cn/)

$$HQ_{i,j} = H_i \times \left(1 - \frac{D_{i,j}^z}{D_{i,j}^z + k^z} \right), \quad (1)$$

where $HQ_{i,j}$ is the HQ for land use type i in pixel j , H_i is the habitat suitability for land use type i , $D_{i,j}$ is the threat level for land use type i in pixel j , k is the half-saturation constant, and z is a default parameter.

Carbon storage

Terrestrial ecosystems absorb CO_2 from the atmosphere through photosynthesis and store it in vegetation, soil, and litter, thereby providing a climate regulating service. The carbon pool includes aboveground biomass, belowground biomass, soil, and dead organic matter [32, 45]. The carbon module of the InVEST tool evaluates CS using the land use data and the carbon density of each land use type. CS was calculated as:

$$C_{total} = C_{above} + C_{below} + C_{soil} + C_{dead}, \quad (2)$$

where C_{total} is the total CS, C_{above} , C_{below} , C_{soil} , C_{dead} are the CS of each land use type in aboveground biomass, belowground biomass, soil, and dead organic matter, respectively.

Soil retention

SR primarily includes erosion control and overland sediment retention, which is particularly important in mountainous regions. The sediment delivery ratio module of the InVEST tool calculates SR by subtracting actual soil erosion from potential soil erosion based on the revised universal soil loss equation [32, 45]. SR was calculated as:

$$SR = R \times K \times LS \times (1 - C \times P), \quad (3)$$

where R is the rainfall erosivity factor, K is the soil erodibility factor, LS is the topographic factor, C is the vegetation practice factor, and P is the management factor.

Water conservation

WC is the ability of ecosystems to retain and store precipitation through their unique structure and water interactions, and to regulate water flows and cycles through evapotranspiration. To evaluate WC, we first used the water yield module of the InVEST tool to calculate water yield by subtracting evapotranspiration from the precipitation based on the regional water balance [32, 45]. Water yield was calculated as:

$$WY = \left(1 - \frac{AET}{P} \right) \times P, \quad (4)$$

where WY is the water yield, AET is the actual annual evapotranspiration, P is the annual precipitation. We then integrated the topographic index, the soil saturation hydraulic conductivity, and the flow rate coefficient to calculate WC as [46]:

$$WC = \min \left(1, \frac{249}{Velocity} \right) \times \min \left(1, \frac{0.9 \times TI}{3} \right) \times \min \left(1, \frac{Ksat}{300} \right) \times WY, \quad (5)$$

where *Velocity* is the flow rate coefficient, *TI* is the topographic index, *Ksat* is the soil saturation hydraulic conductivity, and *WY* is the water yield.

Combined ecosystem service

To assess the spatiotemporal changes in ecosystem dimensions, we standardized the above four ecosystem services by unifying their values between 0 and 1, and then summed them to obtain the combined ecosystem service (CES) [32, 47]:

$$CES = \sum_{i=1}^4 \frac{ES_{i,j} - ES_{i,min}}{ES_{i,max} - ES_{i,min}}, \quad (6)$$

where $ES_{i,j}$ is the value of the ecosystem service i in pixel j , and $ES_{i,max}$ and $ES_{i,min}$ are the maximum and minimum values of the ecosystem service i , respectively.

We calculated the mean value and standard deviation of ecosystem services for each WH site in each year using the R package ‘raster’ (version 3.6-20) [48]. We then used one-way ANOVA to test the difference in the mean value and standard deviation of ecosystem services between karst and non-karst WH sites. Data were checked for normality using the Shapiro–Wilk normality test and log-transformed where necessary.

Trade-off/synergy assessment of ecosystem services

Different from traditional correlation analysis that could only assess trade-offs/synergies between two types of services [33, 49], we used the ranking method to analyze trade-offs/synergies among the four ecosystem services simultaneously [50], which is also widely used in other studies [51–54]. First, we normalized the values using a scale from 0 to 1 by dividing all values by the maximum value for each service. Second, we used the natural break classification method to produce three provision levels (low, medium, and high) for each of the four services (Table S4), which were coded as 1, 2, and 3, respectively. Finally, we determined the interaction codes (*IC*) by combining the provision levels of individual services into an “ecosystem service bundle” using the following formula:

$$IC = HQ \times 1000 + CS \times 100 + SR \times 10 + WC. \quad (7)$$

The output codes were *IC* numbers, which varied between 1111 and 3333. According to the *IC* numbers, we classified the relationships among ecosystem services into strong and weak trade-offs/synergies (Table S5). A strong trade-off occurs when the capacity of one service is high while the capacity of other services is medium or low. A weak trade-off occurs when the capacity of two or three services is high while other services exhibit a low capacity. A high synergy occurs when the capacity of all

services is high or medium. A low synergy occurs when all services exhibit a low or medium capacity. We also used one-way ANOVA to test the difference in the proportion of strong and weak trade-offs/synergies among ecosystem services between karst and non-karst WH sites.

Driving factor analysis of ecosystem services

To identify the key drivers influencing the variation in ecosystem services, we selected a total of 14 potential factors from the domains of climate, topography, landscape, soil, vegetation, and human activities (Table 2). Climate factors included mean annual temperature (MAT) and mean annual precipitation (MAP). Topographic factors included digital elevation model (DEM) and slope. Landscape was indicated by the landscape division index, which has been widely used to describe the degree of landscape fragmentation [47, 55] and was calculated from the land use data using FRAGSTATS software (version 4.2). Soil factors included soil thickness, pH, soil organic carbon (SOC), and bulk density. Vegetation was indicated by the normalized difference vegetation index (NDVI). Human factors included population density, distance from road, nighttime light, and gross domestic product (GDP).

Geographical detector model

GDM is a powerful spatial statistical technique based on the analysis of spatial variation in the geographical layers of variables, which includes factor detection, risk detection, ecological detection, and interactive detection [32, 45, 56, 57]. We used factor detection to quantitatively determine the individual contribution (i.e., explanatory power q -value) of the 14 driving factors to the variation in CES. We also used interactive detection to determine the degree of interaction between any two drivers, as measured by the q -value. The q -value was expressed as follows:

$$q = 1 - \frac{\sum_{h=1}^L N_h \sigma_h^2}{N \sigma^2}, \quad (8)$$

where $h=1, 2, 3, \dots, L$ is the classification or stratification of the driving factor; N_h and σ_h^2 are the sample number and variance of the layer h , respectively; N and σ^2 are the total sample number and variance, respectively; and q is the explanatory power of the driving factor and ranges from 0 to 1. The larger the q value, the greater the contribution of the driving factor to the variation in CES. Interactions between any two factors (e.g., X1 and X2) were classified into five categories based on whether they enhance or diminish the explanatory power when

Table 2 Information on the driving factors of ecosystem services

Factor type	Variable	Description
Climate	MAT	MAT can influence many ecological and biological processes that underpin ecosystem services, such as plant growth, species distribution, evapotranspiration, and decomposition
	MAP	MAP directly affects water availability and associated processes, such as plant growth and distribution, erosion control, and water conservation
Topography	DEM	DEM reflects the terrain and topography of a region, which affects various ecological processes (e.g., water flow and distribution, sediment transport) and human activities
	Slope	Slope determines the speed and direction of surface water flow and the ability to retain soil and resist erosion
Landscape	Landscape division index	A larger landscape division index reflects a greater degree of landscape fragmentation, which reduces ecosystem connection and functions
Soil	Thickness	Soil thickness can affect root penetration, nutrient availability, and water holding capacity, with direct consequences for ecosystem functions
	pH	Soil pH influences nutrient availability and microbial activity, which are crucial to many ecological processes that underpin ecosystem services
	SOC	SOC plays an integral role in soil health, fertility, and texture, directly influencing carbon sequestration, nutrient cycling, erosion control, water retention, and soil biodiversity
	Bulk density	Bulk density reflects soil structure, with consequences for soil aeration, water infiltration and retention, nutrient availability, and plant growth
Vegetation	NDVI	NDVI directly reflects vegetation health and productivity, which can affect multiple services such as carbon sequestration and erosion control
Human	Population density	Population density can reflect the level of human disturbance, which negatively affects ecosystem functions and services
	Distance from road	Road construction can disrupt ecological connection. A long distance from road reflects a low level of human disturbance
	Nighttime light	Nighttime light can disrupt natural light–dark cycles and influence plant physiology and species interactions, with consequences for ecosystem structure and functions
	GDP	GDP reflects economic activities that both depend on and impact ecosystems

Table 3 Classification of factor interaction

Judgment basis	Interaction type
$q(X1 \cap X2) < \min(q(X1), q(X2))$	Non-linear reduction
$\min(q(X1), q(X2)) < q(X1 \cap X2) < \max(q(X1), q(X2))$	Single-factor nonlinearity reduction
$q(X1 \cap X2) > \max(q(X1), q(X2))$	Two-factor enhancement
$q(X1 \cap X2) = q(X1) + q(X2)$	Independent
$q(X1 \cap X2) > q(X1) + q(X2)$	Nonlinear enhancement

acting together, or whether they affect each other independently (Table 3). We conducted the GDM using the R package ‘GD’ (version 1.10), which is an optimal parameters-based GD tool where the optimal parameters are explored as the best combination of spatial data discretization method, break number of spatial strata, and spatial scale parameter [56].

Structural equation modeling

SEM is a widely used tool to identify causal relationships between variables in ecology and environmental science [58–60]. To explore the direct and indirect effects of climate, topography, landscape, soil, vegetation, and

human factors on CES, we constructed a SEM based on a priori conceptual model. We first constructed composite variables for climate, topography, soil, and human factors using their corresponding measured variables, and then included these composite variables in the SEM. We used the following common indicators to check the fitness of the model: (1) Chi-square (χ^2) test, with a $p > 0.05$ indicating that the model has a good fit; (2) root mean square error of approximation (RMSEA), which can overcome the drawback of the overall different value influenced by the estimated parameters in model validation, with a value < 0.05 indicating that the fitness of the model is good; (3) Bentler’s comparative fit index (CFI) and Tucker-Lewis index (TLI), which are between 0 and 1, with a value > 0.95 indicating that the model fits the data very good. We conducted SEM using the maximum-likelihood estimation method with the R package ‘lavaan’ (version 0.6-15) [61].

Results

Spatiotemporal differences in ecosystem services of the WH sites

Overall, there were high spatiotemporal differences in ecosystem services of the four WH sites (Figs. 2, 3, and S1–S7). In terms of spatial variation, the values of

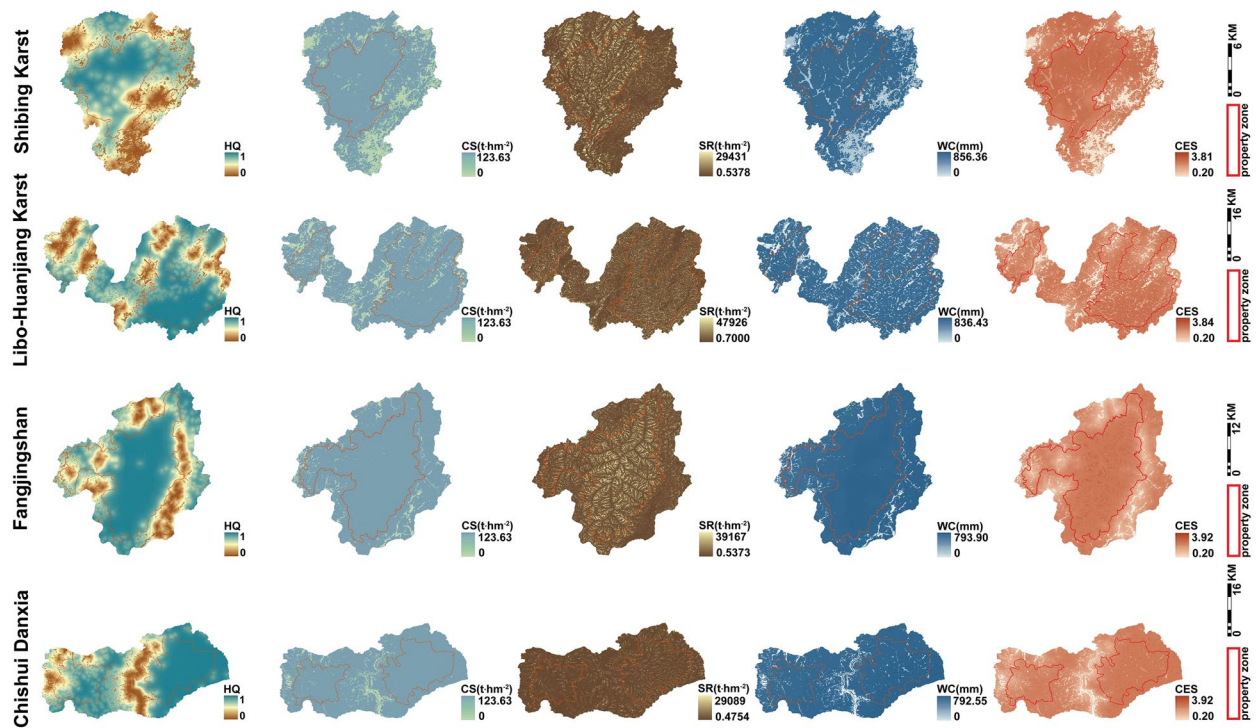


Fig. 2 Spatial differences in ecosystem services of the four World Heritage sites in Southwest China in 2020. HQ: habitat quality; CS: carbon storage; SR: soil retention; WC: water conservation; CES: combined ecosystem service

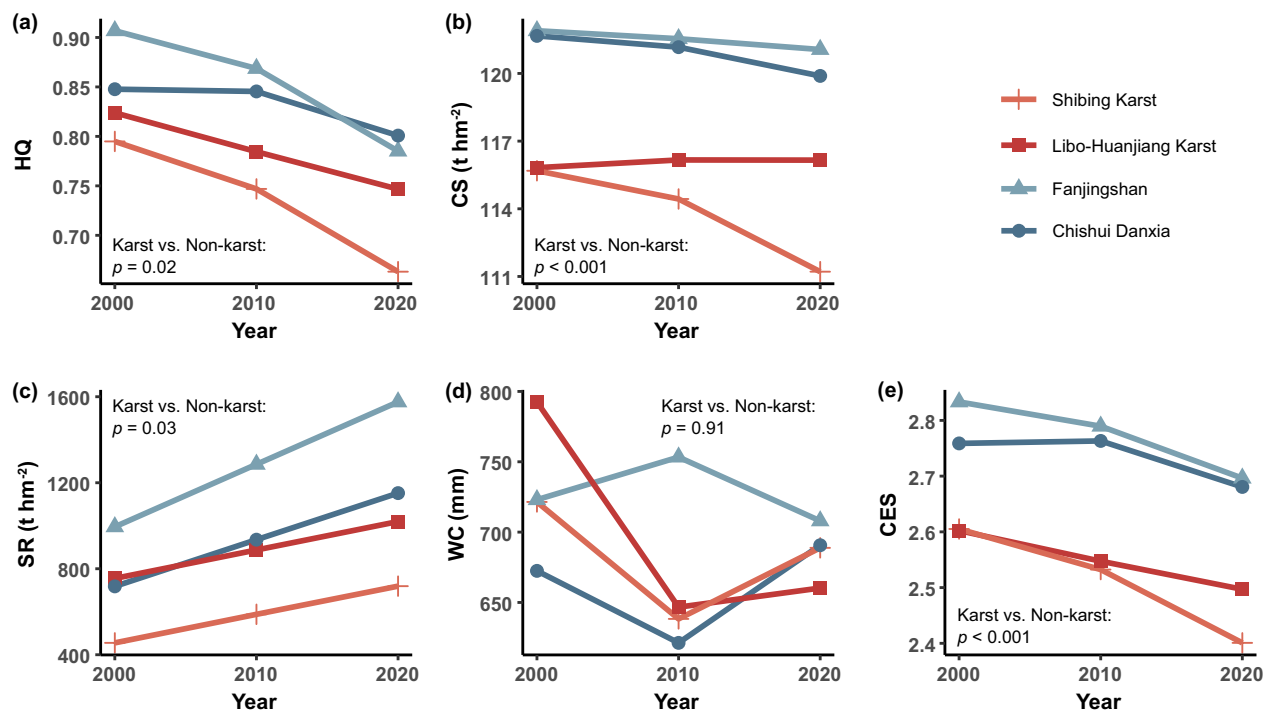


Fig. 3 Temporal variation in the mean value of habitat quality (HQ, a), carbon storage (CS, b), soil retention (SR, c), water conservation (WC, d), and combined ecosystem service (CES, e) of the four World Heritage sites in Southwest China. One-way ANOVA was used to test the difference in the mean value of ecosystem services between karst and non-karst World Heritage sites

ecosystem services in the property zone were much higher than in the buffer zone across the four WH sites (Figs. 2, S1–S4). In terms of temporal variation, irrespective of WH sites, the mean value of HQ gradually decreased from 2000 to 2020 (Fig. 3a), while that of SR increased (Fig. 3c). There was little temporal change in the mean value of CS, except for a decreasing trend over time in Shibing Karst (Fig. 3b). For WC, the mean value decreased from 2000 to 2010 and then increased in 2020 in Chishui Danxia and the two karst WH sites, while the trend was reversed in Fanjingshan (Fig. 3d). Irrespective of WH sites, the mean value of CES gradually decreased from 2000 to 2020 (Fig. 3e). There were similar temporal patterns of ecosystem services between the property zone and the buffer zone of the four WH sites (Figs. S3, S4). Except for WC, the mean values of HQ, CS, and SR were significantly higher in non-karst WH sites than in karst WH sites, resulting in the same pattern for the mean value of CES (Figs. 3, S3, and S4).

There was also temporal variation in the spatial heterogeneity of ecosystem services of the four WH sites (Fig. S5). Specifically, the standard deviation of all ecosystem services increased from 2000 to 2020, except for the standard deviation of CS and WC in Libo-Huanjiang Karst (Fig. S5). These temporal changes in the spatial

heterogeneity of ecosystem services were largely consistent between the property zone and the buffer zone of the four WH sites (Figs. S6, S7). In terms of spatial variation, except for SR, the standard deviation of ecosystem services was greater in the buffer zone than in the property zone of the four WH sites (Figs. S6, S7). Furthermore, compared to non-karst sites, karst WH sites had a significantly higher standard deviation of CES, which was particularly evident in the standard deviation of CS and WC (Figs. S5–S7).

Trade-offs/synergies among ecosystem services of the WH sites

Trade-offs and synergies among ecosystem services varied considerably in space and time (Figs. 4, S8, and S9). In terms of spatial variation, strong trade-offs and weak synergies were mainly distributed in the buffer zone of the four WH sites, whereas weak trade-offs and strong synergies were distributed in both the property and buffer zones (Figs. 4, S8, and S9). In terms of temporal variation, there was little temporal change in the proportion of strong synergies across the four WH sites (Fig. 5a), whereas the proportion of weak synergies increased over time (Fig. 5b). However, the proportion of strong trade-offs decreased from 2000 to 2010 and then increased in

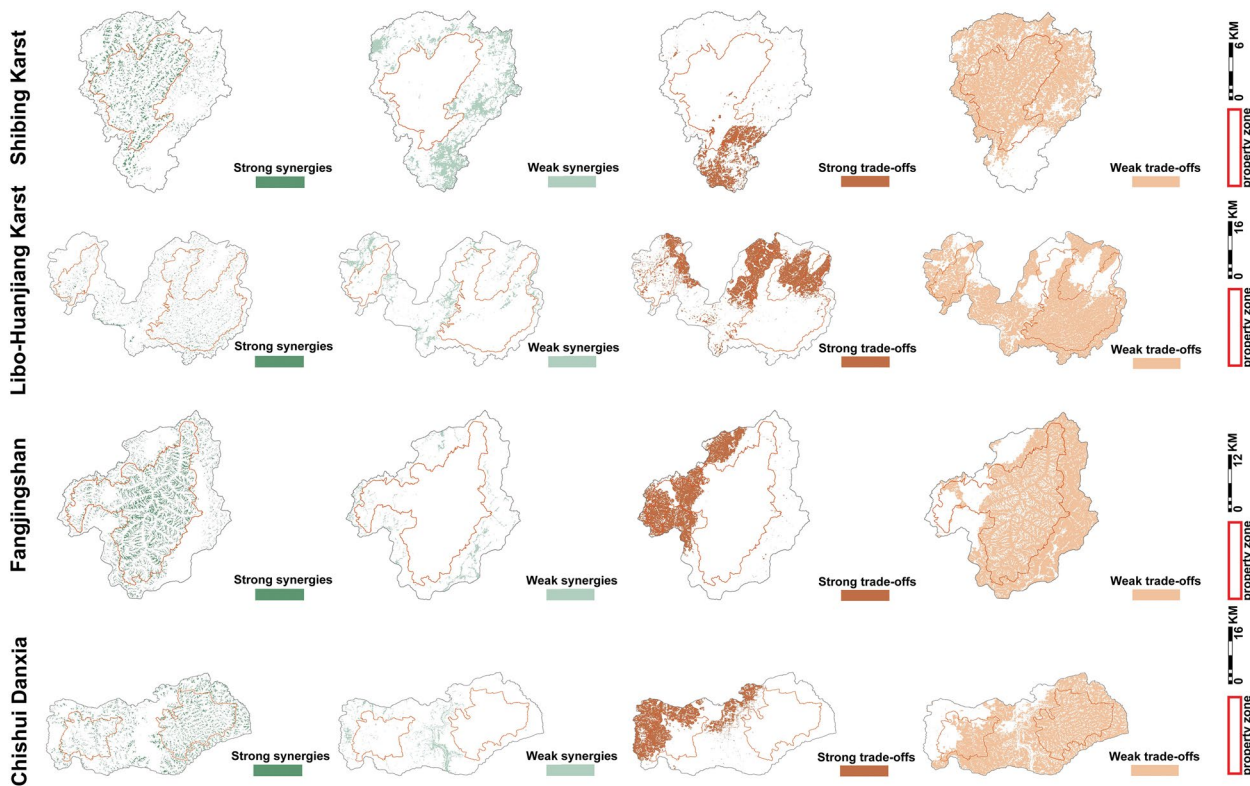


Fig. 4 Spatial differences in trade-offs and synergies among ecosystem services of the four World Heritage sites in Southwest China in 2020

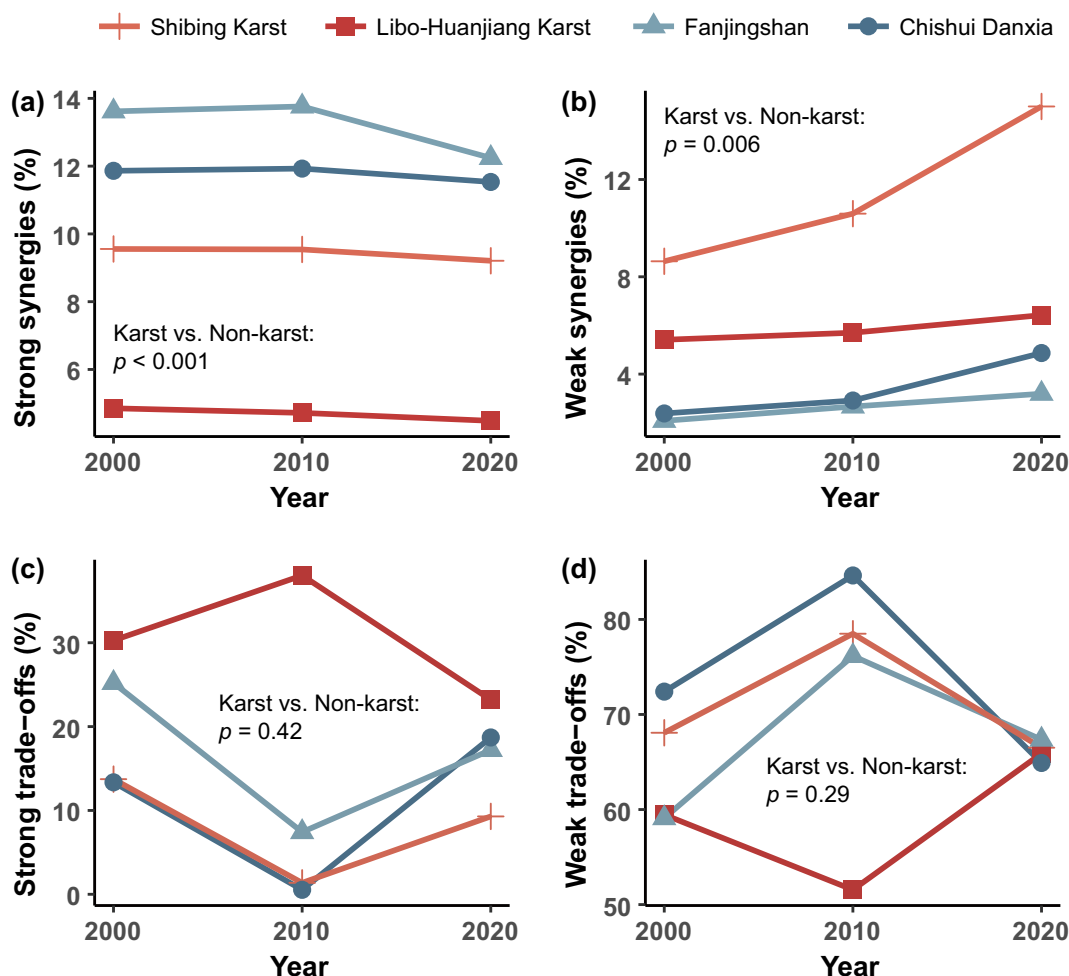


Fig. 5 Temporal variation in the proportion of strong synergies (a), weak synergies (b), strong trade-offs (c), and weak trade-offs (d) among ecosystem services of the four World Heritage sites in Southwest China. One-way ANOVA was used to test the difference in the proportion of trade-offs/synergies between karst and non-karst World Heritage sites

2020 in Shibing Karst and the two non-karst WH sites, while the trend was reversed in Libo-Huanjiang Karst (Fig. 5c). The proportion of weak trade-offs showed an inverse temporal trend compared to that of strong trade-offs (Fig. 5d).

Weak trade-offs among ecosystem services dominated the four WH sites, with a mean proportion of 68.6%, followed by strong trade-offs (15.8%), strong synergies (9.8%), and weak synergies (5.8%) (Fig. 5). Compared to non-karst sites, karst WH sites had a significantly lower proportion of strong synergies and a significantly higher proportion of weak synergies (Fig. 5a, b).

Driving factors of ecosystem services of the WH sites

GDM showed that the 14 driving factors all had significant effects on the variation in CES across the four WH sites ($p < 0.05$), but with different contributions (Fig. 6a). Landscape division index had the largest effect ($q = 0.63$),

followed by NDVI ($q = 0.52$), distance from road ($q = 0.39$), slope ($q = 0.38$), population density ($q = 0.33$), SOC ($q = 0.28$), DEM ($q = 0.25$), soil thickness ($q = 0.22$), bulk density ($q = 0.19$), MAP ($q = 0.14$), GDP ($q = 0.13$), MAT ($q = 0.11$), pH ($q = 0.11$), and nighttime light ($q = 0.05$). The interactive detection results indicated that the interaction between any two factors had a stronger effect than their individual influence, showing either a two-factor enhancement or a nonlinear enhancement (Fig. 6b). Furthermore, higher q -values were primarily found in the interactions with landscape division index, NDVI, distance from road, and population density. The most significant interactions were between landscape division index and NDVI ($q = 0.76$) and between landscape division index and distance from road ($q = 0.74$).

SEM further revealed the causality and direction of different driving factors affecting CES (Fig. 7). Specifically, the landscape division index had a negative effect

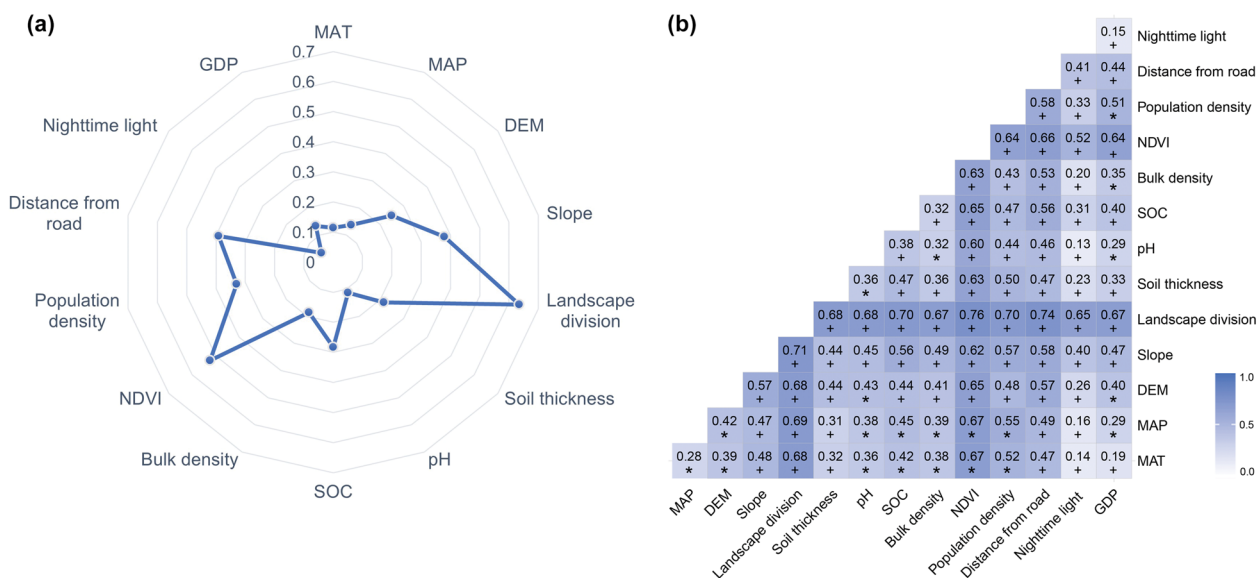


Fig. 6 Results of factor detection (a) and interactive detection (b) for the driving factors influencing the combined ecosystem service of World Heritage sites in Southwest China. '+' denotes a two-factor enhancement, whereas '*' denotes a nonlinear enhancement

on CES, both directly and indirectly through affecting vegetation and soil factors. Topographic factors had a positive effect on CES, both directly and indirectly through affecting vegetation and soil factors. Climate factors had a direct positive effect on CES, but an indirect negative effect through affecting vegetation. Both vegetation and soil factors had a direct positive effect on CES. Human factors, both directly and indirectly

through affecting vegetation, had a positive effect on CES, which was primarily driven by the positive effect of distance from road.

Discussion

General patterns of ecosystem services of the WH sites

Our study showed that the values of ecosystem services were generally higher in the property zone than in the buffer zone of the four WH sites in Southwest China. Given that anthropogenic factors (e.g., distance from road and population density) played an important role in affecting ecosystem services, this result could be attributed to the fact that the property zone is strictly protected, while the buffer zone is often subjected to anthropogenic threats such as agricultural and tourism activities [15]. Across the four WH sites and three periods, the mean values of HQ, CS, SR, and WC were 0.80, 118.06 t hm⁻², 923.78 t hm⁻², and 693.07 mm, respectively, which were generally higher than the mean values of these ecosystem services in non-WH areas [21, 62–65]. This result highlights that WH sites are irreplaceable resources for biodiversity and the basis for ecosystem services [11, 13, 66].

However, our study found that the mean values of HQ and CES decreased from 2000 to 2020 across the four WH sites. Such disappointing reports on WH do not only exist in China but also abound throughout the world in the context of rising threats and declining support [67]. These results suggest that while WH sites are hotspots

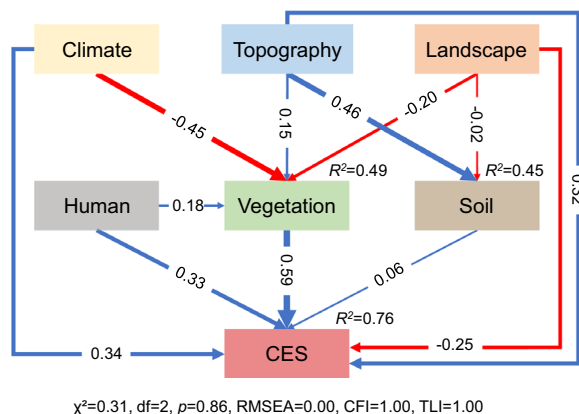


Fig. 7 Structural equation modelling showing the direct and indirect effects of various driving factors on the combined ecosystem service (CES) of World Heritage sites in Southwest China. The model fit indexes show that the model has a very good fitting. Numbers adjacent to arrows are standardized path coefficients, all of which are significant ($p < 0.05$). Arrow width is proportional to the standardized coefficient. Blue and red arrows indicate positive and negative effects, respectively

of biodiversity and ecosystem service provision, they are experiencing a rapid decline in ecosystem services, probably as a result of intensified human activities. For example, we found that cropland area increased between 2000 and 2020 across the four WH sites (Table S6), particularly in Shibing Karst (from 2438.46 to 4187.97 hm²) and Chishui Danxia (from 1626.03 to 3425.94 hm²). Indeed, there is ample evidence that land use conversion from forest to cropland can degenerate ecosystem structure and functions, resulting in declined ecosystem services [21, 32, 68, 69]. On the other hand, the increasing intensity of tourism activities (e.g., a 50-fold increase in the number of tourists in Shibing from 2000 to 2020), and associated construction projects, will inevitably influence the structure and functions of WH ecosystems, thereby decreasing the provision of ecosystem services [14, 16].

Trade-offs and synergies among ecosystem services of the WH sites

In a limited ecosystem, the enhancement of the supply capacity of one ecosystem service is often at the expense of other services [46, 70]. This notion is supported by our result that the four WH sites were dominated by trade-offs among ecosystem services, although they represent the most intact and natural ecosystems with the highest biodiversity on Earth. This finding is also consistent with previous studies reporting that trade-offs among ecosystem services are common [53, 71, 72]. In terms of temporal variation, we found that the proportion of weak synergies generally increased from 2000 to 2020 across the four WH sites. According to our methods, weak synergies meant a low level of at least one service and a lack of services at a high level. Therefore, an increasing proportion of weak synergies indicates the deterioration of ecosystems. This may be because the degeneration of one service can decrease the ability of ecosystems to support other services, resulting in a synergistic decline in ecosystem services [73, 74]. In terms of spatial variation, we found that strong trade-offs and weak synergies were mainly distributed in the buffer zone of the four WH sites. This pattern may result from the intensive human activities that affect ecosystem services and their relationships. As areas with weak synergies were mostly distributed in the buffer zone and increased over time, our study suggests that managers should take timely and effective action in these areas to generate strong synergies among ecosystem services, thereby protecting the outstanding universal value, authenticity, and integrity of WH sites.

Differences in ecosystem services between the karst and non-karst WH sites

Our study showed that non-karst WH sites provided higher levels of HQ, CS, SR, and CES than karst WH sites in Southwest China. This result aligns with previous studies reporting that non-karst areas generally provide higher levels of ecosystem services than karst areas [21, 75, 76]. Furthermore, the spatial heterogeneity of ecosystem services was significantly higher in karst than in non-karst WH sites, which may result from the high landscape fragmentation and heterogeneity that characterize karst ecosystems [77, 78]. Our driving factor analyses also suggested that the landscape division index, an indicator of landscape fragmentation, was an important factor negatively affecting the provision of ecosystem services. Indeed, compared to non-karst landscapes, karst landscapes are more separate and fragmented due to complex water–rock interactions, which can reduce ecosystem connection and functions, thereby limiting the provision of ecosystem services [33, 47, 79]. Furthermore, we found that the interaction between landscape division index and distance from road had the largest effect on ecosystem services. This suggests that intensive human activities could exacerbate the negative impact of landscape fragmentation on ecosystem services of karst WH sites. Therefore, controlling human activities should be emphasized to prevent the decline of ecosystem services in karst WH sites with high landscape fragmentation. While not directly explored in our study, Zhang et al. [21] reported that non-karst areas have higher biodiversity than karst areas. This may also contribute to the higher levels of ecosystem services of non-karst WH sites because high biodiversity can increase the provision of ecosystem services through niche complementarity and sampling effects [80, 81]. Nevertheless, this explanation can also be supported by our results because non-karst WH sites had higher values of HQ, which is widely utilized as a reliable proxy for biodiversity [82–84].

Management implications for WH sites

A clear understanding of spatiotemporal changes and trade-offs/synergies in ecosystem services can have important implications for developing management efforts that balance ecological conservation and socio-economic development in WH sites. Our study showed that the values of ecosystem services were much higher in the property zone where primary forests exist than in the buffer zone where human activities are intensive. This finding highlights that ecological protection should always be given top priority in the development of WH sites. This is particularly important for the management of karst WH sites because karst ecosystems are more

vulnerable to degradation and less likely to recover once degraded due to their special geomorphological and hydrogeological characteristics [8, 9].

Our study also showed that ecosystem services of the WH sites, especially HQ, have continued to decline and areas with weak synergies have continued to increase over the last two decades. These results may suggest that ecological protection and conservation have not been well implemented in the WH sites studied here, which is also indicated by the increased area of cropland (Table S6). Therefore, local governments should set up ecological red lines and persistently implement ecological restoration projects such as Natural Forest Protection Projects, the Grain for Green Program, and the Karst Rocky Desertification Restoration Project to effectively rewild the damaged habitats in WH sites [44]. While prohibiting agricultural activities, managers can also implement community-based conservation policies [85], such as establishing ecological compensation mechanisms and offering positions (e.g., tour guides and law enforcement rangers) to promote the job transition for indigenous residents in WH sites. These solutions could effectively address conflicts between WH management agencies and local communities [44, 66].

Moreover, as increasing tourism activities could be a major threat to the originality and integrity of WH ecosystems, industry transition from traditional tourism to ecotourism is a necessary choice to strike a balance between conservation and development in WH sites [12]. On the one hand, the number of tourists should be controlled so as not to exceed the carrying capacity of WH ecosystems [15, 18]. In this case, strategies that rest attractions and divert tourists can be employed to protect the originality and integrity of WH ecosystems. On the other hand, new projects and infrastructures should be properly planned and assessed to minimize their negative impacts on the outstanding universal value, authenticity, and integrity of WH. Due to the complexity and uniqueness of WH ecosystems, coupled with the protective properties of WH, it is necessary to consider various factors and involve multiple stakeholders to establish a comprehensive and optimal ecotourism strategy for WH sites [16].

Limitations and future directions

Although our study has identified the spatiotemporal patterns and driving factors of ecosystem services between karst and non-karst WH sites in Southwest China, several uncertainties and limitations need to be addressed in future studies. First, due to the special dual (surface and underground) hydrological structure of karst ecosystems, soil, nutrients, and surface water can leak and migrate to other areas along rock fissures,

and sinkholes [2]. However, the InVEST model does not involve surface loss parameters, which may result in higher simulated ecosystem services of karst WH sites than the actual value. Therefore, future studies should incorporate the loss parameters to adjust the model for a more precise simulation. Second, despite the popularity of the InVEST model, it often estimates the annual mean values of ecosystem services without considering seasonal variability and feedback. Furthermore, we only investigated ecosystem services of the four WH sites at three time points. Due to the rapid climate change, ecosystem services may vary considerably between years [86, 87]. Therefore, future studies should evaluate ecosystem services at a higher time resolution to reduce the uncertainty in the temporal pattern. Finally, although the WH sites explored in our study have highly diverse forest types, we estimated ecosystem services based on a coarse land use classification. This may lead to uncertainties in the results because different forest types have different properties (e.g., carbon density, evapotranspiration, and runoff). Therefore, future studies should carefully differentiate vegetation types to improve the accuracy of results for WH sites.

Conclusions

In this study, we investigated the spatiotemporal differences, trade-offs/synergies, and driving factors of ecosystem services between karst and non-karst WH sites in Southwest China and provided relevant suggestions for WH ecosystem management. Importantly, our study showed that ecosystem services varied considerably in space and time, showing an overall higher value in the property zone than in the buffer zone of the WH sites and a decreasing trend over time. These variations in ecosystem services of the WH sites likely resulted from the increased cropland area and intensified tourism activities. Compared to non-karst sites, karst WH sites had significantly lower values of ecosystem services, which were mainly associated with their higher degree of landscape fragmentation and heterogeneity. Taken together, our findings could provide scientific support for decision-making aimed at reconciling conflicts between ecological conservation priorities and economic development demands, and then achieving optimal management and sustainable development of WH sites with different attributes.

Abbreviations

CES	Combined ecosystem service
CFI	Bentler's comparative fit index
CS	Carbon storage
DEM	Digital elevation model
GDM	Geographical detector model
GDP	Gross domestic product

HQ	Habitat quality
MAP	Mean annual precipitation
MAT	Mean annual temperature
NDVI	Normalized difference vegetation index
RMSEA	Root mean square error of approximation
SEM	Structural equation modeling
SOC	Soil organic carbon
SR	Soil retention
TLI	Tucker-Lewis index
WC	Water conservation
WH	World Heritage

Supplementary Information

The online version contains supplementary material available at <https://doi.org/10.1186/s40494-024-01394-z>.

Supplementary Material 1.

Author contributions

KX and YC conceived the study. YC led the analysis, with help from MF and CC. YC led the writing, with substantial input from KX, LR, and CC.

Funding

This study was funded by the Sub-Project of National Key Research and Development Program of China (No. 2022YFF1300703), the Key Science and Technology Program of Guizhou Province (No. 5411 2017 QKHPTRC) and the China Overseas Expertise Introduction Program for Discipline Innovation (No. D17016).

Availability of data and materials

The datasets used and/or analysed during the current study are available from the corresponding author on reasonable request.

Declarations

Competing interests

The authors declare no competing interests.

Author details

¹School of Karst Science, Guizhou Normal University/State Engineering Technology Institute for Karst Desertification Control, Guiyang 550001, China. ²Ministry of Education Key Laboratory for Biodiversity Science and Ecological Engineering, National Observations and Research Station of Wetland Ecosystems of the Yangtze Estuary, Institute of Biodiversity Science and Institute of Eco-Chongming, School of Life Sciences, Fudan University, Shanghai 200438, China.

Received: 7 May 2024 Accepted: 23 July 2024

Published online: 02 August 2024

References

- Sweeting MM. Limestone landscapes of South China. *Geol Today*. 1986;2:11–6.
- Ford DC, Williams PW. *Karst hydrogeology and geomorphology*. Hoboken: John Wiley & Sons, Ltd; 2007.
- Yang MD. The vulnerability of karst environment. *Yunnan Geogr Environ Res*. 1990;2:21–9.
- Yuan DX. Karst in Southwest China and its comparison with karst in North China. *Quat Sci*. 1992;12:352–61.
- UNESCO World Heritage Centre. Operational guidelines for the implementation of the world heritage convention. WHC.21/01. Paris: UNESCO; 2021.
- Xiong KN. Karst landscape and environmental conservation in New Zealand. *J Guizhou Norm Univ*. 1990;13:16–25.
- Xiong KN, Li GC, Wang LY. Study on the protection and sustainable development of South China Karst Libo world natural heritage site. *Chin Gard*. 2012;28:66–71.
- Jiang ZC, Lian YQ, Qin XQ. Rocky desertification in Southwest China: impacts, causes, and restoration. *Earth-Sci Rev*. 2014;132:1–12.
- Chen Y, Xiong KN, Ren XD, Cheng C. Vulnerability comparison between karst and non-karst nature reserves—with a special reference to Guizhou Province, China. *Sustainability*. 2021;13:2422.
- Xiong KN, Chen D, Zhang J, Gu XY, Zhang N. Synergy and regulation of the South China Karst WH site integrity protection and the buffer zone agroforestry development. *Herit Sci*. 2023;11:218.
- Allan JR, Venter O, Maxwell S, Bertzky B, Jones K, Shi Y, Watson JEM. Recent increases in human pressure and forest loss threaten many Natural World Heritage Sites. *Biol Conserv*. 2017;206:47–55.
- Fei GY, Xiong KN, Fei GH, Zhang HP, Zhang SR. The conservation and tourism development of World Natural Heritage sites: the current situation and future prospects of research. *J Nat Conserv*. 2023;72:126347.
- Luo L, Wang HJ, Chen ZC, Wang XY, Guo HD. Biodiversity co-benefits of World Heritage protection. *Innov Life*. 2024;2:100051.
- You WB, Ji ZR, Wu LY, Deng XP, Huang DH, Chen BR, Yu JA, He D. Modeling changes in land use patterns and ecosystem services to explore a potential solution for meeting the management needs of a heritage site at the landscape level. *Ecol Indic*. 2017;73:68–78.
- Zhang J, Xiong KN, Liu ZJ, He LX. Research progress on world natural heritage conservation: its buffer zones and the implications. *Herit Sci*. 2022;10:102.
- Yang XS, Mu DJ, Hu MJ, Xiao SZ, Meng JN. Establishment of the management effectiveness assessment system of world natural heritage and empirical analysis—a case study of Fanjingshan. *Herit Sci*. 2023;11:124.
- Wuepper D, Patry M. The World Heritage list: which sites promote the brand? A big data spatial econometrics approach. *J Cult Econ*. 2017;41:1–21.
- Selcuk O, Karakas H, Cizel B, Ipekci Cetin EI. How does tourism affect protected areas?: a multi-criteria decision making application in UNESCO natural heritage sites. *Nat Hazards*. 2023;117:1923–44.
- Costanza R, d'Arge R, deGroot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill RV, Paruelo J, et al. The value of the world's ecosystem services and natural capital. *Nature*. 1997;387:253–60.
- Byrd KB, Flint LE, Alvarez P, Casey CF, Sleeter BM, Soulard CE, Flint AL, Sohl TL. Integrated climate and land use change scenarios for California rangeland ecosystem services: wildlife habitat, soil carbon, and water supply. *Landsc Ecol*. 2015;30:729–50.
- Zhang SH, Xiong KN, Qin Y, Min XY, Xiao J. Evolution and determinants of ecosystem services: insights from South China karst. *Ecol Indic*. 2021;133:108437.
- Zhang SH, Xiong KN, Min XY, Zhang S. Demographic shrinkage promotes ecosystem services supply capacity in the karst desertification control. *Sci Total Environ*. 2024;917:170427.
- Cord AF, Bartkowski B, Beckmann M, Ditrlich A, Hermans-Neumann K, Kaim A, Lienhoop N, Locher-Krause K, Priess J, Schroter-Schlaack C, et al. Towards systematic analyses of ecosystem service trade-offs and synergies: Main concepts, methods and the road ahead. *Ecosyst Serv*. 2017;28:264–72.
- Wang T, Chen XD, Zheng X, Lu YY, Han F, Yang ZP. Identification of priority conservation areas for natural heritage sites integrating landscape ecological risks and ecosystem services: a case study in the Bogda, China. *Int J Environ Res Public Health*. 2022;19:2044.
- Sui H, Liu C, Ye C, Xu X, Sui T. Analysis of ecosystem resilience in Jiuzhaigou Valley Scenic Area under the effect of geohazards. *Front Earth Sci*. 2023;10:1053327.
- Wang Z, Yang ZP, Shi H, Han F, Liu Q, Qi JW, Lu YY. Ecosystem health assessment of world natural heritage sites based on remote sensing and field sampling verification: Bayanbulak as case study. *Sustainability*. 2020;12:2610.
- Wang KL, Zhang CH, Chen HS, Yue YM, Zhang W, Zhang MY, Qi XK, Fu ZY. Karst landscapes of China: patterns, ecosystem processes and services. *Landsc Ecol*. 2019;34:2743–63.
- Xiong KN, Fu YL, Chen H, Du FJ, Xiao SZ, et al. South China Karst (Phase II) for World Heritage Nomination: Guilin Karst, Shibing Karst, Jinfoshan Karst and Huanjiang Karst. Beijing: Ministry of Housing and Urban-Rural Development, the People's Republic of China; 2013.

29. Xiong KN, Liang YN, Zhao CY, Li YH, Xie SY, et al. South China Karst for World Heritage Nomination: Shilin Karst (Yunnan), Libo Karst (Guizhou) and Wulong Karst (Chongqing). Beijing: Ministry of Housing and Urban-Rural Development, the People's Republic of China; 2005.
30. Xiong KN, Chen H, Rong L, Ying B, Du FJ, Xiao SZ, et al. Fanjingshan for world heritage nomination. Beijing: Ministry of Housing and Urban-Rural Development, the People's Republic of China; 2016.
31. Xiong KN, Xiao SZ, Chen H, Rong L, Peng J, Du FJ. World heritage and Chishui Danxia. Beijing: Higher Education Press; 2012.
32. Fang LL, Wang LC, Chen WX, Sun J, Cao Q, Wang SQ, Wang LZ. Identifying the impacts of natural and human factors on ecosystem service in the Yangtze and Yellow River Basins. *J Clean Prod.* 2021;314:127995.
33. Liu SJ, Wang ZJ, Wu W, Yu LF. Effects of landscape pattern change on ecosystem services and its interactions in karst cities: a case study of Guiyang City in China. *Ecol Indic.* 2022;145:109646.
34. Bai LM, Xiu CL, Feng XH, Liu DQ. Influence of urbanization on regional habitat quality: a case study of Changchun City. *Habitat Int.* 2019;93:102042.
35. Feng Z, Jin XR, Chen TQ, Wu JS. Understanding trade-offs and synergies of ecosystem services to support the decision-making in the Beijing-Tianjin-Hebei region. *Land Use Pol.* 2021;106:105446.
36. Ouyang X, Tang LS, Wei X, Li YH. Spatial interaction between urbanization and ecosystem services in Chinese urban agglomerations. *Land Use Pol.* 2021;109:123851.
37. Tang F, Wang L, Guo YQ, Fu MC, Huang N, Duan WS, Luo M, Zhang JJ, Li W, Song W. Spatio-temporal variation and coupling coordination relationship between urbanisation and habitat quality in the Grand Canal, China. *Land Use Pol.* 2022;117:106119.
38. Deng CX, Liu JY, Liu YJ, Li ZW, Nie XD, Hu XQ, Wang LX, Zhang YT, Zhang GY, Zhu DM, Xiao LH. Spatiotemporal dislocation of urbanization and ecological construction increased the ecosystem service supply and demand imbalance. *J Environ Manage.* 2021;288:112478.
39. Gu YY, Lin NF, Ye X, Xu MJ, Qiu J, Zhang K, Zou CX, Qiao XN, Xu DL. Assessing the impacts of human disturbance on ecosystem services under multiple scenarios in karst areas of China: insight from ecological conservation red lines effectiveness. *Ecol Indic.* 2022;142:109202.
40. Li YM, Yang X, Wu BW, Zhao JZ, Jiang WX, Feng XJ, Li YT. Spatio-temporal evolution and prediction of carbon storage in Kunming based on PLUS and InVEST models. *PeerJ.* 2023;11:e15285.
41. Liang J, Li S, Li XD, Li X, Liu Q, Meng QF, Lin AQ, Li JJ. Trade-off analyses and optimization of water-related ecosystem services (WRESs) based on land use change in a typical agricultural watershed, southern China. *J Clean Prod.* 2021;279:123851.
42. Wang JZ, Zhang Q, Gou TJ, Mo JB, Wang ZF, Gao M. Spatial-temporal changes of urban areas and terrestrial carbon storage in the Three Gorges Reservoir in China. *Ecol Indic.* 2018;95:343–52.
43. Yang D, Liu W, Tang LY, Chen L, Li XZ, Xu XL. Estimation of water provision service for monsoon catchments of South China: applicability of the InVEST model. *Landsc Urban Plan.* 2019;182:133–43.
44. Chen X, Yu L, Cao Y, Xu YD, Zhao ZC, Zhuang YB, Liu XH, Du ZR, Liu T, Yang B, et al. Habitat quality dynamics in China's first group of national parks in recent four decades: evidence from land use and land cover changes. *J Environ Manag.* 2023;325:116505.
45. Yang J, Zhai DL, Fang Z, Alatalo JM, Yao Z, Yang W, Su Y, Bai Y, Zhao G, Xu J. Changes in and driving forces of ecosystem services in tropical south-western China. *Ecol Indic.* 2023;149:110180.
46. Yang JY, Li JS, Fu G, Liu B, Pan LB, Hao HJ, Guan X. Spatial and temporal patterns of ecosystem services and trade-offs/synergies in Wujiang River Basin, China. *Remote Sens.* 2023;15:4099.
47. Ma S, Wang LJ, Wang HY, Zhang X, Jiang J. Spatial heterogeneity of ecosystem services in response to landscape patterns under the Grain for Green Program: a case-study in Kaihua County, China. *Land Degrad Dev.* 2022;33:1901–16.
48. Hijman RJ, van Etten J, Sumner M, Cheng J, Baston D, Bevan A, Bivand R, Busetto L, Canty M, Fasoli B, et al. raster: geographic data analysis and modeling. R package version 3.6-20. 2023.
49. Zhang ZY, Liu YF, Wang YH, Liu YL, Zhang Y, Zhang Y. What factors affect the synergy and tradeoff between ecosystem services, and how, from a geospatial perspective? *J Clean Prod.* 2020;257:120454.
50. Carr MH, Zwick PD. Smart land-use analysis—the LUCIS model. Redlands: Esri Press; 2008.
51. Cademus R, Escobedo FJ, McLaughlin D, Abd-Elrahman A. Analyzing trade-offs, synergies, and drivers among timber production, carbon sequestration, and water yield in *Pinus elliotii* forests in Southeastern USA. *Forests.* 2014;5:1409–31.
52. Jafarzadeh AA, Mahdavi A, Shamsi SRF, Yousefpour R. Assessing synergies and trade-offs between ecosystem services in forest landscape management. *Land Use Pol.* 2021;111:105741.
53. Wang JY, Cao Y, Fang XQ, Li GY, Cao Y. Identification of the trade-offs/synergies between rural landscape services in a spatially explicit way for sustainable rural development. *J Environ Manage.* 2021;300:113706.
54. Wang YC, Li BY. Dynamics arising from the impact of large-scale afforestation on ecosystem services. *Land Degrad Dev.* 2022;33:3186–98.
55. McGarigal KS, Cushman S, Neel MC, Ene E. FRAGSTATS: spatial pattern analysis program for categorical maps. 2002.
56. Song YZ, Wang JF, Ge Y, Xu CD. An optimal parameters-based geographical detector model enhances geographic characteristics of explanatory variables for spatial heterogeneity analysis: cases with different types of spatial data. *GISci Remote Sens.* 2020;57:593–610.
57. Dong W, Wu X, Zhang JJ, Zhang YL, Dang H, Lu YH, Wang C, Guo JY. Spatiotemporal heterogeneity and driving factors of ecosystem service relationships and bundles in a typical agrapastoral ecotone. *Ecol Indic.* 2023;156:111074.
58. Cebrían-Piqueras MA, Karrasch L, Kleyer M. Coupling stakeholder assessments of ecosystem services with biophysical ecosystem properties reveals importance of social contexts. *Ecosyst Serv.* 2017;23:108–15.
59. Leong RAT, Fung TK, Sachidhanandam U, Drillet Z, Edwards PJ, Richards DR. Use of structural equation modeling to explore influences on perceptions of ecosystem services and disservices attributed to birds in Singapore. *Ecosyst Serv.* 2020;46:101211.
60. Hu BA, Wu HF, Han HR, Cheng XQ, Kang FF. Dramatic shift in the drivers of ecosystem service trade-offs across an aridity gradient: evidence from China's Loess Plateau. *Sci Total Environ.* 2023;858:159836.
61. Rosseel Y. lavaan: an R package for structural equation modeling. *J Stat Softw.* 2012;48:1–36.
62. Han HQ, Dong Y. Assessing and mapping of multiple ecosystem services in Guizhou Province, China. *Trop Ecol.* 2017;58:331–46.
63. Niu L, Shao Q. Soil conservation service spatiotemporal variability and its driving mechanism on the Guizhou Plateau, China. *Remote Sens.* 2020;12:2187.
64. Zuo LY, Gao JB, Du FJ. The pairwise interaction of environmental factors for ecosystem services relationships in karst ecological priority protection and key restoration areas. *Ecol Indic.* 2021;131:108125.
65. Xu GY, Xiong KN, Shu T, Shi YJ, Chen LS, Zheng LL, Fan HX, Zhao ZM, Yang ZH. Bundling evaluating changes in ecosystem service under karst rocky desertification restoration: projects a case study of Huaijiang-Guanling, Guizhou province, Southwest China. *Environ Earth Sci.* 2022;81:302.
66. Fu YY, Xiong KN, Zhang ZZ. Ecosystem services and ecological compensation of world heritage: a literature review. *J Nat Conserv.* 2021;60:125968.
67. Osipova E, Shadie P, Zwahlen C, Osti M, Badman T. IUCN world heritage outlook 2: a conservation assessment of all natural world heritage sites. Gland: IUCN; 2017. p. 92.
68. Yang YJ, Wang K, Liu D, Zhao XQ, Fan JW. Effects of land-use conversions on the ecosystem services in the agro-pastoral ecotone of northern China. *J Clean Prod.* 2020;249:119360.
69. Zhong LN, Wang J, Zhang X, Ying LX. Effects of agricultural land consolidation on ecosystem services: trade-offs and synergies. *J Clean Prod.* 2020;264:121412.
70. Dai EF, Wang XL, Zhu JJ, Zhao DS. Methods, tools and research framework of ecosystem service trade-offs. *Geogr Res.* 2016;35:1005–16.
71. Zhang JJ, Zhu WB, Zhu LQ, Li YH. Multi-scale analysis of trade-off/synergistic effects of forest ecosystem services in the Funiu Mountain Region, China. *J Geogr Sci.* 2022;32:981–99.
72. Ding LL, Liao Y, Zhu CM, Zheng QW, Wang K. Multiscale analysis of the effects of landscape pattern on the trade-offs and synergies of ecosystem services in Southern Zhejiang Province, China. *Land.* 2023;12:949.
73. Bennett EM, Peterson GD, Gordon LJ. Understanding relationships among multiple ecosystem services. *Ecol Lett.* 2009;12:1394–404.
74. Raudsepp-Hearne C, Peterson GD, Bennett EM. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proc Natl Acad Sci USA.* 2010;107:5242–7.

75. Wang B, Yang ST, Lue CW, Zhang J, Wang YJ. Comparison of net primary productivity in karst and non-karst areas: a case study in Guizhou Province, China. *Environ Earth Sci*. 2010;59:1337–47.
76. Han HQ, Liu Y, Gao HJ, Zhang YJ, Wang Z, Chen XQ. Tradeoffs and synergies between ecosystem services: a comparison of the karst and non-karst area. *J Mt Sci*. 2020;17:1221–34.
77. Cheng C, Gao M, Zhang YD, Long MZ, Wu YJ, Li XN. Effects of disturbance to moss biocrusts on soil nutrients, enzyme activities, and microbial communities in degraded karst landscapes in southwest China. *Soil Biol Biochem*. 2021;152:108065.
78. Cheng C, Chen Y, Yu CY, Wu JJ, Zhu SX, Long MZ, Li XN. Moss biocrusts buffer soil CO₂ effluxes in a subtropical karst ecosystem. *CATENA*. 2023;232:107388.
79. Xia H, Kong W, Zhou G, Sun OJ. Impacts of landscape patterns on water-related ecosystem services under natural restoration in Liaohe River Reserve, China. *Sci Total Environ*. 2021;792:148290.
80. Isbell F, Calcagno V, Hector A, Connolly J, Harpole WS, Reich PB, Scherer-Lorenzen M, Schmid B, Tilman D, van Ruijven J, et al. High plant diversity is needed to maintain ecosystem services. *Nature*. 2011;477:199–202.
81. Cheng C, Liu ZK, Song W, Chen X, Zhang ZJ, Li B, van Kleunen M, Wu JH. Biodiversity increases resistance of grasslands against plant invasions under multiple environmental changes. *Nat Commun*. 2024;15:4506.
82. Tang F, Fu MC, Wang L, Zhang PT. Land-use change in Changli County, China: predicting its spatio-temporal evolution in habitat quality. *Ecol Indic*. 2020;117:106179.
83. Yohannes H, Soromessa T, Argaw M, Dewan A. Spatio-temporal changes in habitat quality and linkage with landscape characteristics in the Beressa watershed, Blue Nile basin of Ethiopian highlands. *J Environ Manage*. 2021;281:111885.
84. Chen X, Yu L, Du ZR, Xu Y, Zhao JY, Zhao HL, Zhang GL, Peng DL, Gong P. Distribution of ecological restoration projects associated with land use and land cover change in China and their ecological impacts. *Sci Total Environ*. 2022;825:153938.
85. Balint PJ. Improving community-based conservation near protected areas: the importance of development variables. *Environ Manage*. 2006;38:137–48.
86. Weiskopf SR, Rubenstein MA, Crozier LG, Gaichas S, Griffis R, Halofsky JE, Hyde KJW, Morelli TL, Morisette JT, Munoz RC, et al. Climate change effects on biodiversity, ecosystems, ecosystem services, and natural resource management in the United States. *Sci Total Environ*. 2020;733:137782.
87. Moss ED, Evans DM, Atkins JP. Investigating the impacts of climate change on ecosystem services in UK agro-ecosystems: an application of the DPSIR framework. *Land Use Pol*. 2021;105:105394.

Publisher's Note

Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.