RESEARCH PAPER

SOIL EROSION VERSUS SOIL RETENTION CAPACITY: AN IMPACT ASSESSMENT OF REGULATING ECOSYSTEM SERVICE PROVISION IN IRAN

Kazem Nosrati[1,*], Maram Hassan[1], Alireza Salehipour Milani[1], Mostafa Keshtkar[2], Somaiyeh Khaleghi[1], Adrian L. Collins[3]

[1]Department of Physical Geography, Faculty of Earth Sciences, Shahid Beheshti University, 1983969411 Tehran, Iran
[2]Department of Planning and Designing the Environment, Environmental Science Research Institute, Shahid Beheshti University, Tehran, Iran
[3]Rothamsted Research, North Wyke, Okehampton, EX20 2SB, UK
***Corresponding author:** k_nosrati@sbu.ac.ir
Received: August 22nd 2024 / Accepted: November 12nd 2025 / Published: December 31st 2025
https://doi.org/10.24057/2071-9388-2025-3768

ABSTRACT. The objective of this study was to determine the impact of erosion on soil retention as an ecosystem service and its relationship with soil quality in a mountainous catchment in Iran. In this regard, 42 soil samples were collected from rangelands, rainfed, and irrigated farming areas. Thirteen physical and chemical soil attributes were measured. Principal component analysis was applied to identify a soil quality index (SQI). The respective ranges of soil erosion rates from rangelands, rainfed farming lands, and irrigated farming lands were estimated to be 0.2 – 46.4, 0.18 – 0.20, and 0.00 – 0.18 t ha^{-1} yr^{-1}. The SQI estimates ranged between 3.2 – 4.0 for the rangelands compared with corresponding estimates of 4.0 – 5.7 for the rainfed farming lands and 5.7 – 8.4 for the irrigated farming lands. Soil retention was estimated to range between 0 – 0.01 t ha^{-1} yr^{-1} (average = 0.005 t ha^{-1} yr^{-1}) for rangelands, 0.01 – 0.03 t ha^{-1} yr^{-1} (average = 0.02 t ha^{-1} yr^{-1}) for rainfed farmlands, and 0.03 – 3.5 t ha^{-1} yr^{-1} (average = 1.8 t ha^{-1} yr^{-1}) for irrigated farming lands. Negative relationships were observed among soil erosion, soil quality, and soil retention, emphasising the sensitivity of soil quality to the soil erosion rates estimated for different land use types. This study provides evidence for the negative effects of soil erosion under different land uses regarding the degradation of soil quality and soil retention as an ecosystem service.

KEYWORDS: Soil loss; soil quality index, soil degradation; soil retention

CITATION: Nosrati K., Hassan M., Milani A. S., Keshtkar M., Khaleghi S., Collins A. L. (2025). Soil Erosion Versus Soil Retention Capacity: An Impact Assessment Of Regulating Ecosystem Service Provision In Iran. Geography, Environment, Sustainability, 4 (18), 80-91
https://doi.org/10.24057/2071-9388-2025-3768

ACKNOWLEDGEMENTS: We acknowledge the support of Grant number 600.871 funded by the research council of Shahid Beheshti University, Tehran, Iran. The contribution to this work by ALC was funded by UKRI-BBSRC (UK Research and Innovation-Biotechnology and Biological Sciences Research Council) via grant award BB/X010961/1 (Resilient Farming Futures) – specifically work package 2 - BBS/E/RH/230004B; Detecting agroecosystem 'resilience' using novel data science methods.

Conflict of interests: The authors reported no potential conflict of interests.

INTRODUCTION

Increasing economic development and the resulting land use change have accelerated soil erosion. This is now documented as one of the most severe environmental challenges globally (Wijesundara et al., 2018). Importantly, soil is a non-renewable resource that requires improved management for sustainability given population growth (Lal, 2015). Soil erosion heavily impacts anthropogenically-modified ecosystems, including managed forests and croplands (Obidike-Ugwu et al., 2025). Accelerated erosion frequently causes other issues such as sealing, acidification, salinisation, alkalisation, diffuse pollution, and biodiversity decline (Ayoubi et al., 2014; Nabiollahi et al., 2018; Nosrati and Collins, 2019). Soil erosion is a major issue that adversely impacts environmental quality and food security, and it also reduces crop productivity (Rahmanipour et al., 2014). To mitigate the specific damages caused by soil erosion, such as decreased soil depth and organic matter, and soil compaction leading to reduced fertility and productivity, fertilisers and pesticides are used widely (Emadodin et al., 2012; Nosrati and Van Den Eeckhaut, 2012). These are used to sustain crop yield, but these applications harm human and environmental health (Amuah et al., 2024). Approximately 70% of Iran's drylands have exhibited signs of desertification, with croplands experiencing the highest risk (Eskandari Dameneh et al., 2021). The economic ramifications are significant, with the total cost of soil and water degradation, along with fertiliser use in agriculture, estimated at around US $12.8 billion. This is approximately 4% of Iran's total GDP and about 35% of the agricultural sector's GDP (Emadodin et al., 2012).

To quantify soil quality, it is necessary to measure the physical, chemical, and biological attributes that affect soil processes, functions, and services (Dominati et al., 2010; Dominati, 2013). The properties and various factors affecting soils must be evaluated to assess soil quality (Aziz et al., 2009), as soil quality is a fundamental basis for ecosystem functions. Soil quality can be described as the capacity of soil, under a particular land use or within a given ecosystem, to maintain fertility and environmental quality, and to enhance the health and diversity of plants, animals, and micro-organisms (Dilly et al., 2018; Karlen et al., 1997). Therefore, the physical, chemical, and biological properties and functions of soil are consistent in both soil-related ecosystem services, such as soil retention, and soil quality indicators (Aitkenhead and Coull, 2019; Black et al., 2010; Van Eekeren et al., 2010).

The soil ecosystem provides many services (Gómez-Baggethun et al., 2010). These benefits vary both spatially and temporally (Fisher et al., 2009). Soil ecosystem services are classified into four categories: provisioning services (e.g., provision of medicines, building materials, and nutrients), regulating services (e.g., water regulation, water quality control, erosion control and soil retention, greenhouse gas storage/retention), supporting services (e.g., nutrient cycling, water cycling, biodiversity), and cultural services (e.g., recreational activities, cultural heritage, aesthetic experience, spiritual enrichment) (Comerford et al., 2013; Paul et al., 2021). Among these services, the regulating category ensures environmental safety and sustainability (Bo-Jie et al., 2004). Soil erosion degrades regulating ecosystem services. As a result, controlling soil erosion can restore those specific soil-related ecosystem services (Steinhoff-Knopp et al., 2021).

Soil-related ecosystem services can be evaluated using the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) tool for ecosystem services mapping (Mousazadeh et al., 2018). Within this tool, soil erosion is assessed using the universal soil loss equation (USLE) model (Zhang et al., 2019). Tools such as InVEST offer the opportunity to map soil retention in areas where such conceptual frameworks have not yet been applied, thereby providing critical new information for informing soil management. On this basis, the work reported in this paper applied InVEST to a study area in Iran. The Zar-Abad catchment is a mountainous case study area that has experienced progressive human development. It is characterised by steep slopes, different land use types, and lithology vulnerable to erosion. Collectively, these factors have resulted in accelerated soil erosion rates and subsequent sedimentation downstream. The objective of this study was to investigate the impact of soil erosion on soil retention as an indicator of soil ecosystem regulating services, and its relationship with soil quality in the Zar-Abad catchment, Iran. To this end, the study addressed the following research questions: (i) Do soil quality indices significantly differ across various land use types? (ii) What is the relationship between soil erosion, soil retention, and soil quality?

MATERIALS AND METHODS

Study area and data sources

Our study was conducted in the Zar-Abad catchment (36^0_46″ to 36^0_56″N and 50^0_32″ to 50^0_50″E). This area is part of the Alamout drainage basin, located in the southwest of the Alborz Mountains, 139 km northwest of Tehran, Iran (Fig. 1). The Zar-Abad catchment covers an area of 110 km^2 and includes rangelands, irrigated orchards, and rainfed farms. Rangelands make up the majority of the current land cover, accounting for 45%. Elevations range from 1000 m to over 3000 m. Slopes exceed 20% in 70% of the study area. The average annual rainfall between 2007 and 2018 was approximately 399 mm. Based on data from the Iranian Meteorological Organisation, the study area has a cold and mountainous climate. The mean daily temperatures from 2007 to 2018 were -8.1 °C in the coldest month and 35.5 °C in the warmest month.

Geologically, the study area is underlain by sandstone, red and green marl, basalt, salt, shale, tuff, and young terraces. The predominant geological formation consists of silt and sand combined with red and green marl.

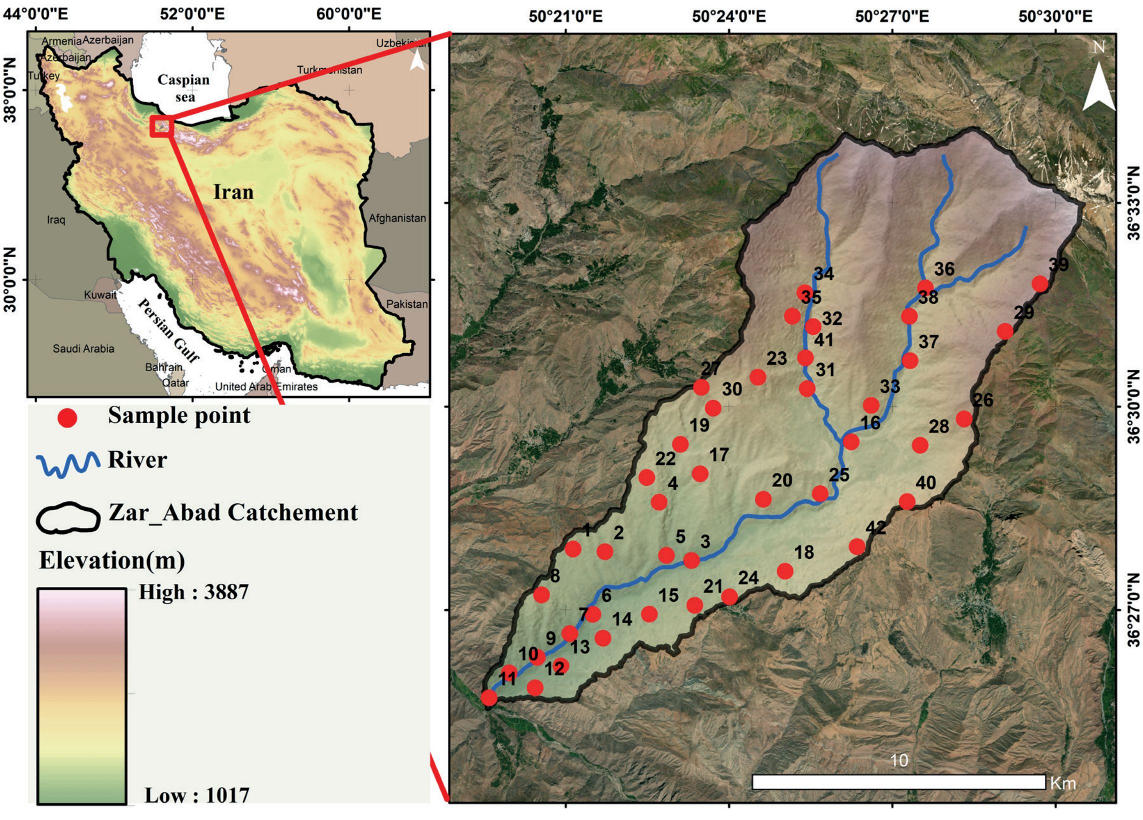

Fig. 1. The geographical location of the Zar-Abad study catchment, and soil sampling locations

The average annual rainfall and the monthly rainfall for the period 2007 to 2018 were used as meteorological data in this study. The land use map and land cover classification were extracted from Sentinel 2 image data with a 10 m spatial resolution. We also used a digital elevation model (DEM) with 12.5 m resolution taken from ALOS PALSAR images downloaded from the Vertex Alaska website (https://search.asf.alaska.edu/#/). Soil data were obtained from field sampling and laboratory analyses. Figure 2 shows the flowchart outlining the methodological steps and input data in this study.

A sampling approach was determined based on the geology and types of land use in the study area, resulting in a total of 42 soil samples being collected. These comprised 17 samples from rangelands, 15 from rainfed farming lands, and 10 from irrigated farming (orchards) lands. The samples were taken to a depth of 0-20 cm, bearing in mind that only the surface layers are at severe risk of erosion (Nosrati and Collins, 2019). Each soil sample weighed approximately 1 kg. Following field collection, all soil samples were dried and sieved using a 2 mm sieve. A set of soil physico-chemical attributes was analysed for each sample. These included absolute particulate size distribution (sand, silt, and clay), organic matter (OM) content, electrical conductivity (EC), pH, water holding capacity (WHC), saturation percentage (SP), available water content (AWC), bulk density (BD), particle density (PD), as well as carbonate calcium ($CaCO_3$), potassium (K), sodium (Na) and phosphorus (P) content. These were selected to represent critical soil quality indicators. More details on the specifics of the soil properties used in this study are presented in Table 1.

Estimation of soil erosion and soil retention

The InVEST model

The soil retention services in the InVEST software calculate the ecosystem's capacity for soil retention. This is done by considering the maximum potential soil loss and potential soil loss (Sharp et al., 2014) according to the following equations:

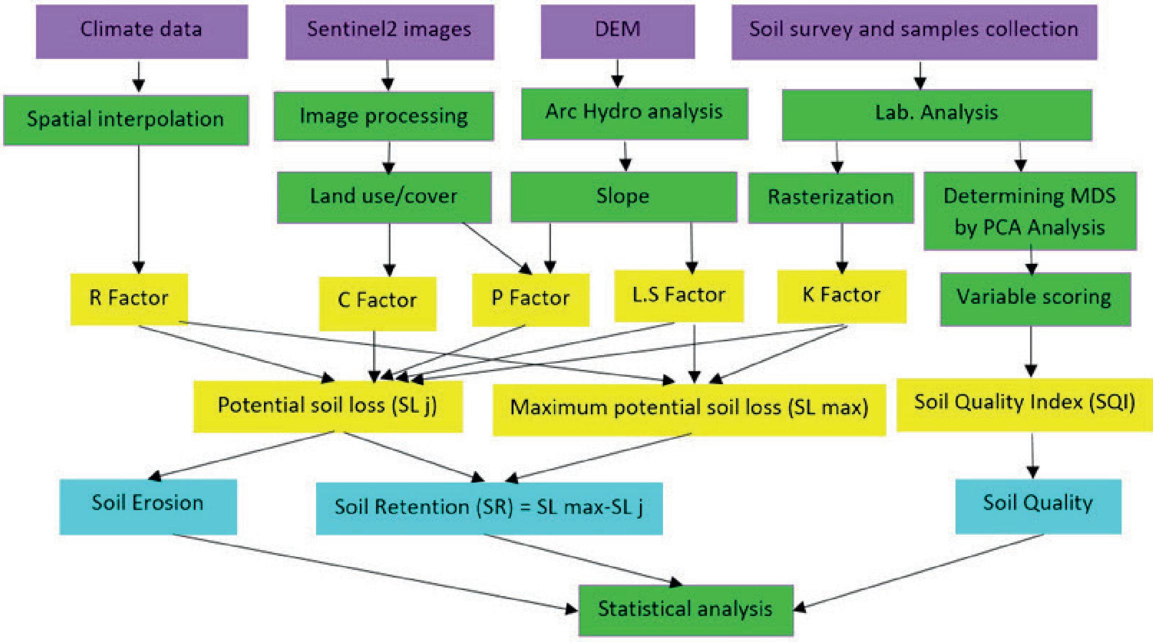

Fig. 2. Flowchart outlining the methodological steps and input data

Table 1. Descriptions of the soil parameters used in this study

Soil Factor	Unit	Method	References
Soil Texture (clay, silt, and sand)	%	Hydrometer method	Kroetsch and Wang (2008)
SP	%	Weight method	Carter and Gregorich (2007)
AWC	%	Pressure plate extractor	Carter and Gregorich (2007)
WHC	%	Pressure plate extractor	Carter and Gregorich (2007)
BD	$Mg\ m^{-3}$	Core method	Palmer et al. (2002)
PD	$Mg\ m^{-3}$	Pycnometer procedure	Blake and Hartge (1986)
EC	dsm^{-1}	Saturated soil _paste extract	Corwin and Lesch (2003)
pH		Saturated soil _paste extract	Robbins and Wiegand (1990);
OM	%	Walkley and Black	Walkley and Black (1934)
$CaCO_3$	%	Calcimetre method	Şenlikci et al. (2015)
K and Na	$mg\ kg^{-1}$	Flame photometric method	Helmke and Sparks (1996)
P	$mg\ kg^{-1}$	Spectrophotometer method	Gburek et al. (2000)

$$Soil\ retention\ (SR) = SLmax\,(R \times K \times L \times S) - SLj\,(USLE) \tag{1}$$

$$SLmax = R \times K \times LS \tag{2}$$

$$SLj = R \times K \times LS \times C \times P \tag{3}$$

where SLmax is the maximum potential soil loss regardless of the vegetation factor. SLj denotes the potential soil erosion that can be calculated from the universal soil loss equation (USLE). R, K, LS, C, and P are the rainfall erosivity, the soil erodibility, the slope-length and steepness, the crop management, and the conservation practice factors, respectively in the USLE model (Irvem et al., 2007; Wischmeier and Smith, 1978). The USLE factors are determined as described immediately below.

Rainfall erosivity factor (R)

The term rain erosion was proposed by Wischmeier (1978) to describe the effect of climate on soil erosion. Based on rainfall data for the Zar-Abad station, this study calculated R-values for the period 2007 to 2018 using average annual rainfall (Renison et al., 2010). Accordingly, the Fournie (F) Index (Eq. 4) was used with the average annual precipitation. Where the F-index was <55 mm, Equation 5 was applied, and where it exceeded 55 mm, Equation 6 was used (Renard et al., 2011).

$$R = \sum_{k=1}^{n12} \frac{pi^2}{p} \tag{4}$$

if: $$F < 55mm;\ R = \left(0.07397 \times F^{1.847}\right) \tag{5}$$

if: $$F \geq 55mm;\ R = \left(95.77 - 6.081 \times F + F + 0.477 \times F^2\right) \tag{6}$$

where: pi is the monthly rainfall (mm) and p represents the annual rainfall (mm).

Soil erodibility factor (K)

The soil erodibility factor (t ha h ha^{-1} MJ^{-1} mm^{-1}) measures how sensitive soil particles are to detachment and transport by rainfall and runoff (Sun et al., 2014). Based on soil texture, 24 samples (57% of the 42 samples) were clay textured, with texture being an important factor controlling erodibility (Jong, 1994). Wischmeier (1978) developed the following concept based on laboratory analyses (Eq. 7):

$$K = \left(\left(2.1 \times 10^{(-4)\,(12-a)} \times M^{(1.14)}\right) + (3.25\,(b-2)) + \frac{2.5\,(c-3)}{759}\right) \tag{7}$$

where M represents silt (%) + very fine sand (%) × (100 – clay (%)); a is OM content (%), and b is the soil structure code, where 1 = very fine granular [1–2 mm], 2 = fine granular [2–5 mm], 3 = medium or coarse granular [5–10 mm], and 4 = blocky, platy, or massive [>10 mm]. Finally, c represents the soil profile permeability, where 1 is high, 2 is moderate to high, 3 is moderate, 4 is moderate to slow, 5 is slow, and 6 is very slow.

Slope length and steepness factor (LS)

Slope length and steepness (LS) factors represent the impact of topography and morphology on the rate of soil erosion. As the slope increases, the cumulative runoff and the velocity of surface runoff also increase (Jong, 1994). Methods for calculating the L and S factors using GIS_SAGA can be found within the SAGA programme. This was used with our DEM for the study area.

The LS factor in the USLE shows the combined effect of slope length and slope steepness on soil erosion. Moore and Wilson (1992) provided a simplified method for calculating the LS factor, which is particularly useful for GIS-based calculations. This method focuses on the influence of slope length (L) and slope steepness (S) on soil erosion (Badora and Wawer, 2023). Both SAGA GIS and ArcGIS have tools for calculating the LS factor. They typically use DEMs as input and implement algorithms for flow direction and accumulation to determine slope length and steepness.

The LS factor is calculated using the following formula (Moore and Wilson, 1992):

$$LS = \left(\frac{Slope\ length}{22.1}\right)^{0.4} \times \left(\frac{\sin\theta \times 0.01745}{0.0896}\right) \times 1.4 \tag{8}$$

where Slope Length is Flow accumulation multiplied by Cell resolution (DEM), and θ is Slope in degrees. In this study, the LS factor was calculated using digital elevation models (DEMs) with 12.5 m resolution obtained from ALOS PALSAR images downloaded from the Vertex Alaska website (https://search.asf.alaska.edu/#/) to derive slope and flow path information in GIS_SAGA.

Crop management factor (C)

The crop management factor is one of the most sensitive spatial and temporal controls of soil erosion. It depends on variations in plant growth stages, vegetation types, and rainfall (Zhang et al., 2010). The C-factor quantifies the effect of vegetation cover and management on soil loss in USLE/RUSLE, ranging from approximately 0 (full protection) to 1 (bare soil) (Renard, 1997; Wischmeier, 1978). To map the C-factor, we used Sentinel-2 surface reflectance imagery. The Normalized Difference Vegetation Index (NDVI) was computed from the near-infrared (Band 8) and red (Band 4) using ENVI 5.3 software (Eq. 9), as follows:

$$NDVI = \frac{NIR - RED}{NIR + RED} \tag{9}$$

A cloud-free image acquired during the main erosive season of the study year was selected to represent protective ground cover when erosion risk is most relevant in RUSLE applications (Ayalew et al., 2020). We acknowledge that multi-date NDVI composites reduce temporal noise; however, the single-date approach is widely used when time-series stacks are unavailable (Ayalew et al., 2020).

NDVI values were then transformed to C using the exponential function proposed for continental-scale erosion mapping (Eq. 10).

$$C = exp\left(-\alpha \frac{NDVI}{\beta - NDVI}\right)^{\beta} \tag{10}$$

where α and β shape the NDVI–C curve. More specifically, Van Leeuwen and Sammons (2004) considered 2.5 and 1, which has been shown to provide robust results across diverse environments when using MODIS-NDVI data.

Conservation practice factor (P)

The P-factor represents the ratio of soil loss with a given conservation practice compared to that under conventional tillage up-and-down slope (Wischmeier and Smith, 1978). In this study, land use/land cover (LULC) classes were derived from Sentinel-2 MSI imagery using an object-based image analysis (OBIA) approach in eCognition Developer 64. OBIA allows for the segmentation and classification of homogeneous land parcels and reduces spectral confusion that often occurs in pixel-based classifications (Blaschke, 2010). The LULC map was validated with field surveys and high-resolution Google Earth images to ensure classification accuracy.

The slope gradient was obtained from the 12.5 m ALOS PALSAR DEM. The classified LULC and slope layers were spatially overlaid in a GIS environment. P-factor values were assigned to each LULC–slope combination following the guidelines of Wischmeier (1978) and updated values provided in subsequent studies (Kouli et al., 2009; Lufafa et al., 2003; Phinzi et al., 2021). This procedure allowed the parameterisation of the P-factor by integrating remote sensing data, topographic information, and empirical lookup tables. This provided a transparent and reproducible methodology.

Soil quality index

The application of the soil quality index approach has recently been expanded to quantify soil chemical, physical, and biological indicators that affect the soil's ability to function effectively. In the study reported herein, fifteen different factors affecting soil quality were measured. Our soil quality index (SQI) was calculated using the three steps described below:

(1) To identify the SQI, a small set of soil characteristics representing the so-called minimum data set (MDS) was selected (Fang et al., 2024; Garrigues et al., 2013). The MDS for SQI can be selected based on expert opinion (Andrews et al., 2004) or statistical analyses (Rojas et al., 2016). In our study, the MDS was selected using the load and eigenvalue obtained from principal component analysis (PCA). PCA is considered one of the most common (Doran and Parkin, 1994; Wang et al., 2021) and flexible methods for identifying the MDS for SQI (Juhos et al., 2016). Based on the PCA (Doran and Safley, 1997), the soil properties with the highest factor loading (absolute value) in each PC were shortlisted for the MDS (Andrews et al., 2004).

(2) Using expert opinion (Andrews and Carroll, 2001), the indicators in the MDS were normalised based on a standard scoring function: optimal is better, more is better, and less is better (Si in Equation 10). Therefore, all attributes were converted into a 0-1 range value. Two techniques can be used here: linear or non-linear scoring. In this study, linear scoring was used. For the 'more is better' approach, the values were ranked in rising order and each case divided by the highest observation. For the 'less is better' approach, the values were ranked in declining order, with the lowest value divided by each observation.

(3) For each property in the MDS, the ratio of explained variance for each principal component to the total variance explained by all principal components (total cumulative variance) (Wi in Equation 11) was calculated. On this basis, the SQI was calculated for each soil sample using Equation 11:

$$SQI = \sum_{i=1}^{n=i} Wi \times Si \times 10 \tag{11}$$

where Wi and Si are the weights and scoring rank of the soil attribute selected for the MDS, respectively. Marzaioli et al. (2010) divided the SQI into three classes: low soil quality (SQI < 0.55), medium soil quality (0.55 < SQI < 0.70), and high soil quality (SQI > 0.70). The index value was multiplied by 10 to provide index values in a range of 1 to 10 rather than 0 to 1, as this has been found to be more understandable for producers and other users (Andrews et al., 2004).

Statistical analyses

To assess differences in the SQI across various land use types, a one-way analysis of variance (ANOVA) was conducted. Subsequently, Scheffé's post hoc test was employed to identify statistically homogeneous subsets among the land use categories (Landau and Everitt, 2003). The relationships among soil quality, soil erosion, and soil retention were examined using correlation analysis. All analyses were conducted using SPSS version 22.

RESULTS AND DISCUSSION

Soil erosion estimates based on the USLE

The average annual rainfall data were used to compute the average annual R factor values, which ranged from 616 to 2207 MJ mm ha^{-1} h^{-1} yr^{-1} (Fig. 3A). The results of the K factor indicated that soil erodibility ranged between 0.004 and 0.28 t ha h ha^{-1} MJ^{-1} mm^{-1}. Here, most areas have high erodibility in the upstream portions of the Zar-Abad catchment, whereas erodibility decreases downstream (Fig. 3B). The value of the LS factor ranged within 0–99%. In general, the increase in the length and slope percentage due to the resultant intensification of the velocity and strength of surface flow increased the amount of soil erosion per unit area (Fig. 3°C). The amount of vegetation varied between 0.1–0.6 (values closer to 1 indicate denser vegetation cover and vice versa) (Fig. 3D). The conservation practice values (P-factor; values between 0.1 and 1.0) were determined based on protective operations: a value of 0.1 for orchard farming lands with terraces and gabion check dams and 1.0 for lands without any soil protection operations (Fig. 3E). The results indicated that the rate of soil erosion ranged between 0 and 46.4 t ha^{-1} yr^{-1}. The respective ranges of the soil erosion rates for rangelands, rainfed farming lands, and irrigated farming lands (orchards) were estimated as 0.2–46.4, 0.18–0.2, and 0.0–0.18 t ha^{-1} yr^{-1}.

Soil retention

The potential soil loss and maximum potential soil loss were estimated to be in the ranges of 0 and 46.4 t ha^{-1} yr^{-1}, and 9.0 to 46.4 t ha^{-1} yr^{-1}, respectively (Fig. 4A). Therefore, the corresponding range for soil retention in the study area was 0 to 3.5 t ha^{-1} yr^{-1} (Fig. 4B). The estimates of soil retention for the rangelands, rainfed farming lands, and irrigated farming lands (orchards) ranged between 0–0.01 t ha^{-1} yr^{-1} (with a mean value of 0.005 t ha^{-1} yr^{-1}), 0.01–0.03 t ha^{-1} yr^{-1} (with a mean value of 0.02 t ha^{-1} yr^{-1}), and 0.03–3.5 t ha^{-1} yr^{-1} (with a mean value of 1.8 t ha^{-1} yr^{-1}).

Soil quality index (SQI)

Changes in soil quality can be estimated to evaluate the impacts of different land uses and their corresponding management practices (Arshad and Martin, 2002). Table 2 summarises the results of using PCA to determine the MDS. Five components were calculated to have an

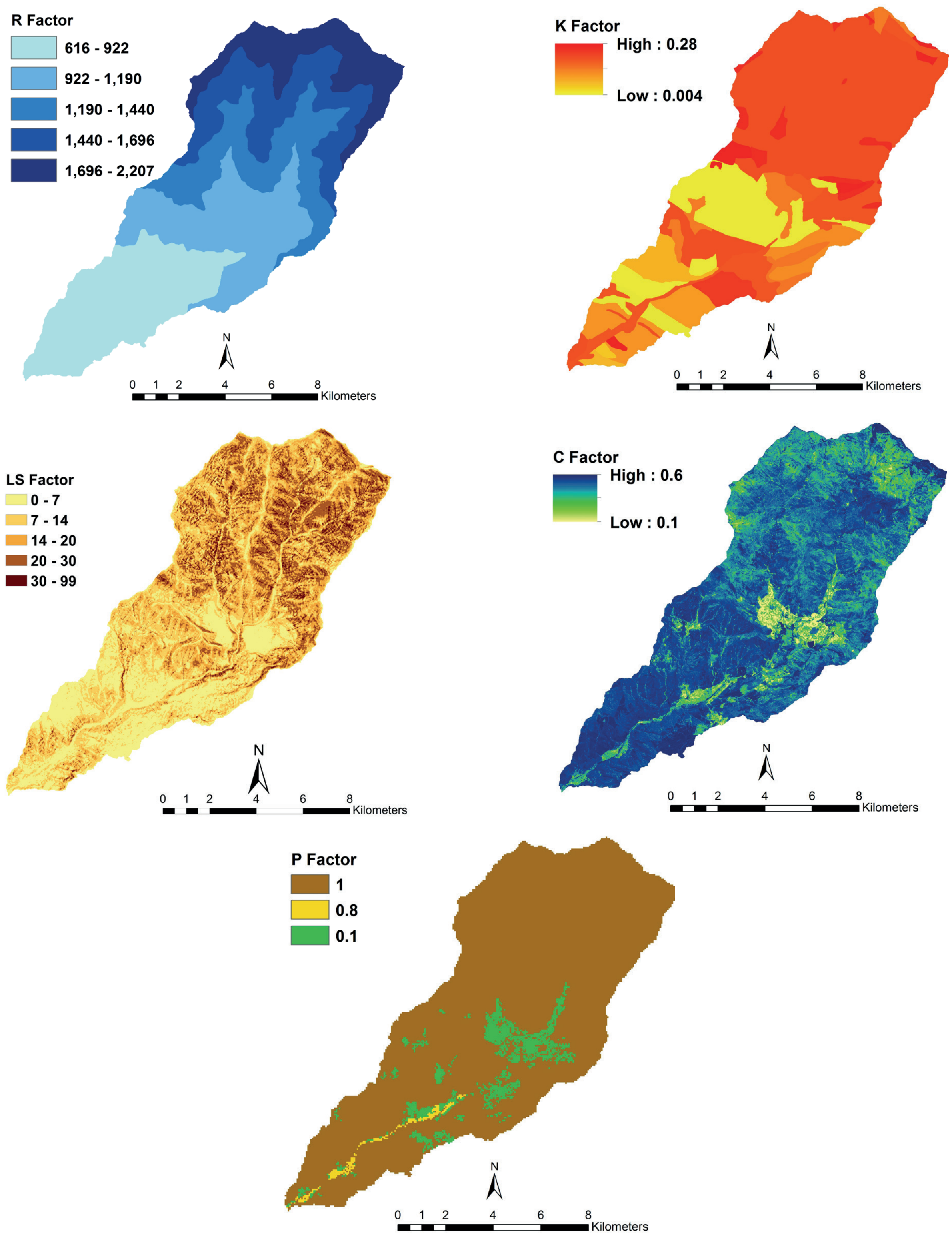

Fig. 3. Maps of the factors for soil erosion: rainfall erosivity factor (R) (MJ mm ha^{-1} h^{-1} yr^{-1}), soil erodibility factor (K) (t ha h ha^{-1} MJ^{-1} mm^{-1}), slope length and steepness factor (LS), D) crop management factor (C), and conservation practice factor (P)

eigenvalue >1 and can therefore be used in the MDS. The cumulative variance was 68.8%. Table 2 reports the results of the rotating components matrix, in which the loading coefficients of the factors were defined for each component. The highest PCs loadings were considered a condition for selecting the final MDS, which comprised OM, EC, P, sand content and K (Table 2).

A 'more is better' scoring approach was used for P, K, and OM (Andrews et al., 2004; Marzaioli et al., 2010; Rahmanipour et al., 2014). In contrast, a 'less is better' scoring function was employed for EC (Derakhshan-Babaei et al., 2021; Nabiollahi et al., 2018; Rahmanipour et al., 2014). The sand content followed an 'optimal r function (Davari et al., 2020; Derakhshan-Babaei et al., 2021; Rahmanipour et al., 2014). The final results indicated that sand content, EC, K, P, and OM had the highest to lowest weights, respectively (Eq. 12):

$$SQI = 0.35\left(S_{Sand}\right) - 0.24\left(S_{EC}\right) + 0.15\left(S_{K}\right) + 0.13\left(S_{P}\right) + 0.11\left(S_{OM}\right) \tag{12}$$

Using Equation 11, the soil quality index ranged between 3.15 and 8.40 (SQI values are in a range of 1 to 10), indicating that soil quality increased from the upstream to the downstream parts of the Zar-Abad catchment. This pattern

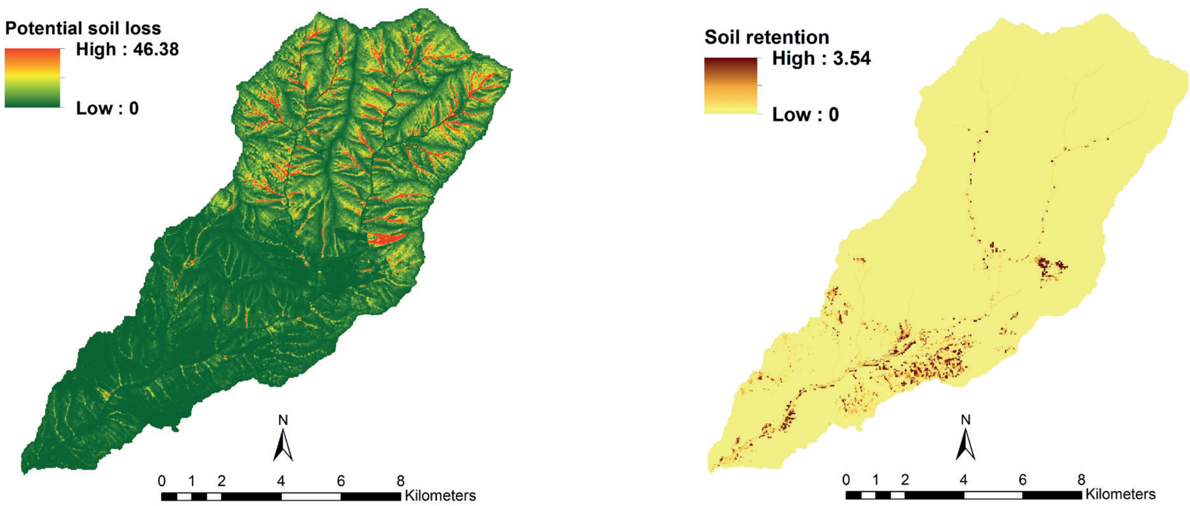

Fig. 4. A) Maps of potential soil loss (t ha^{-1} yr^{-1}) and B) soil retention (t ha^{-1} yr^{-1})

Table 2. The results of principal component analysis (PCA) in selecting the minimum data set (MDS)

Variables	PC1	PC2	PC3	PC4	PC5	Mean	Standard Deviation
Sand (%)	**.74**	-0.10	-0.56	-0.19	0.08	44.40	19.94
Silt (%)	-.28	0.35	0.59	-0.07	0.25	32.15	23.97
Clay (%)	-.73	-0.06	0.36	0.26	-0.22	23.44	10.57
SP (%)	-.07	-0.01	0.68	0.50	-0.24	32.72	14.73
AWC (%)	.02	0.21	-0.11	-0.66	-0.26	9.60	2.13
WHC (%)	.23	0.27	0.15	0.59	0.02	15.67	7.48
BD (Mg m^{-3})	.70	-0.11	-0.02	-0.15	-0.18	1.8	0.3
PD (Mg m^{-3})	0.08	-0.02	-0.02	-0.01	-0.16	2.10	0.45
EC (dS m^{-1})	-0.14	**0.86**	-0.22	0.16	0.13	0.67	0.19
pH	-0.08	0.80	0.08	-0.14	-0.21	7.72	0.26
OM (%)	0.15	0.50	-0.01	-0.20	**-0.57**	2.83	1.41
CaCO3 (%)	-0.70	0.41	0.03	-0.11	-0.04	9.97	6.46
K (g kg^{-1})	-0.08	-0.31	**-0.82**	-0.03	0.08	15.76	3.99
Na (g kg^{-1})	0.44	0.18	0.32	0.31	-0.36	19.19	4.14
P (g kg^{-1})	-0.25	-0.10	-0.08	**0.67**	-0.01	14.50	9.54
Eigenvalue	3.58	2.50	1.64	1.424	1.19		
% Total variance	23.85	16.65	10.91	9.50	7.93		
% Cumulative variance	23.85	40.50	51.41	60.91	68.84		

PC, principal component. The bold value corresponds to the selected attribute in each PC used to calculate the soil quality index (SQI).

reflects the different intrinsic conditions and management factors affecting soil erosion, and consequently soil quality and structure. The SQI ranged between 3.2 and 4.0 for the rangelands compared with corresponding estimates of 4.0 to 5.7 for the rainfed farming lands and 5.7 to 8.4 for the irrigated farming lands.

The relationship between soil erosion, soil retention, and soil quality is fundamental to understanding soil health. Soil erosion is the process by which the top layer of soil is removed by natural forces like wind and water. This removal can lead to a loss of valuable topsoil, which is rich in organic matter and nutrients. Soil retention, conversely, refers to the ability of the soil to stay in place, resisting erosive forces. Factors such as vegetation cover, soil structure, and topography play crucial roles in soil retention. When soil is eroded, the remaining soil often experiences a decline in its quality. This is because the most fertile and nutrient-rich components are typically lost first. Consequently, reduced soil quality can manifest as lower organic matter content, decreased water-holding capacity, and a deficiency in essential plant nutrients. This degradation makes the soil less suitable for agriculture and can negatively impact ecosystems. Conversely, effective soil retention helps to preserve soil quality. Maintaining a stable soil structure, with adequate vegetation and organic matter, enhances its resistance to erosion. Healthy soil, which is well-retained, supports robust plant growth, which in turn further improves soil structure and its ability to retain moisture and nutrients. Therefore, promoting soil retention is a key strategy for safeguarding soil quality and its long-term productivity.

The one-way ANOVA results confirmed that soil quality, as a dependent variable, showed differences across land use types, the independent variables, at the 0.05 confidence level. The significance level was 0.02. Pairwise comparisons from the post hoc test indicated that soil quality in irrigated farming and rangelands was significantly different at the 0.05 level. However, the pairwise comparison of irrigated farming and rainfed farming showed no significant difference (Table 3). There was also no significant difference between the SQI of rangelands and rainfed farming at the 0.05 level (Table 3). Figure 5 demonstrates that the SQI in irrigated farming was higher than in rainfed and rangeland areas, with the latter showing the lowest values.

The relationship between soil erosion, soil quality, and soil retention indicated that the rate of soil erosion decreased from the upstream to the downstream parts of the Zar-Abad study catchment (Fig. 6). This is because the upstream areas of the catchment are very susceptible to soil erosion. The main issues in this region are steep slopes, a lack of vegetation cover, and the erosive impact of rainfall, all of which degrade the chemical and physical components of the soil. Downstream areas in the study catchment experience less soil erosion due to more extensive vegetation cover (irrigated farming, orchards), gentler slopes, lower precipitation intensity, and consequently, lower surface runoff. Therefore, soil quality and soil retention improve from upstream to downstream (Fig. 6).

Table 4 presents the correlations between soil quality, soil retention, and soil erosion. Soil quality shows the strongest correlation with soil retention at the 0.01 confidence level. The relationship between soil quality and soil erosion is a strong negative correlation, as is the relationship between soil retention and soil erosion. Therefore, as soil quality and soil retention increase, the soil erosion rate decreases sharply (Table 4).

Overall, the results indicated that soil quality ranged between 3.15 and 8.40, with a decreasing trend from the downstream to the upstream portions of the study catchment. The estimated soil retention ranged between 0–3.54 and increased from the upstream to the downstream portions of the study catchment. This is due to the reduction in slope and rainfall, as well as increased vegetation density, associated with the transition from rangeland to orchards and rainfed land use types. In general, soil retention and soil quality exhibit the same pattern, in contrast to the pattern of soil erosion (Fig. 6).

The estimated erosion rates in this case study, when compared to soil erosion rates estimated for other catchments in Iran, suggest that our findings are reasonable for the study area. For example, the annual soil erosion in most drainage basins in Iran was estimated to be between 7.5 and 25 t ha-1 yr-1 using empirical methods by Jalalian et al. (1994). Afshar et al. (2010), using the 137Cs method, estimated the gross erosion rate and net soil deposition in western Iran to be 29.8 t ha^{-1} yr^{-1} and 21.8 t ha^{-1} yr^{-1}, respectively. Studies have shown that the Zar-Abad catchment experiences high soil loss when compared to some studies globally and within Iran. According to Wuepper et al. (2020), the global average soil erosion rate is 2.4 t ha^{-1} yr^{-1}. However, rates can vary from less than 1 t ha^{-1} yr^{-1} in some regions to over 20 t ha^{-1} yr^{-1} in others. The spatial distribution of soil erosion in the Zar-Abad catchment, located in the southern Alborz mountains, showed high soil erosion, particularly in rangelands and on steep slopes. Therefore, the steep slopes of the Alborz rangelands have a high potential for soil erosion. This aligns with studies such as that by Doulabian et al. (2021), who noted that the highest soil erosion is expected in the western and northern regions of Iran. Similarly, Mohammadi et al. (2021) stated that the average annual soil erosion in Iran is 16.5 t ha^{-1}, with the highest soil loss values occurring in the north, west, and southwest parts of Iran, and on the steep slopes of the Alborz and Zagros mountains. Ebrahimi et al. (2021) explained that annual soil

Table 3. Results of the one-way ANOVA using the post hoc Scheffe test comparing the SQI between different land uses

Land use (i)	Land use (j)	p- value
Irrigated farming	Rangeland	0.02*
	Rainfed farming	0.43
Rangeland	Rainfed farming	0.31

* The p-value is statistically significant (F = 4.7, p = 0.02).

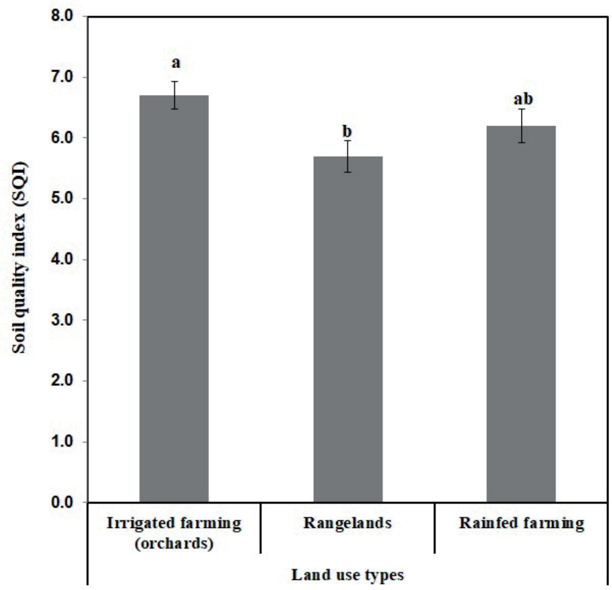

Fig. 5. The SQI values calculated for different land uses in the study area. Error bars represent standard errors. Letters (a, b, c) above the bars indicate statistically significant differences between groups

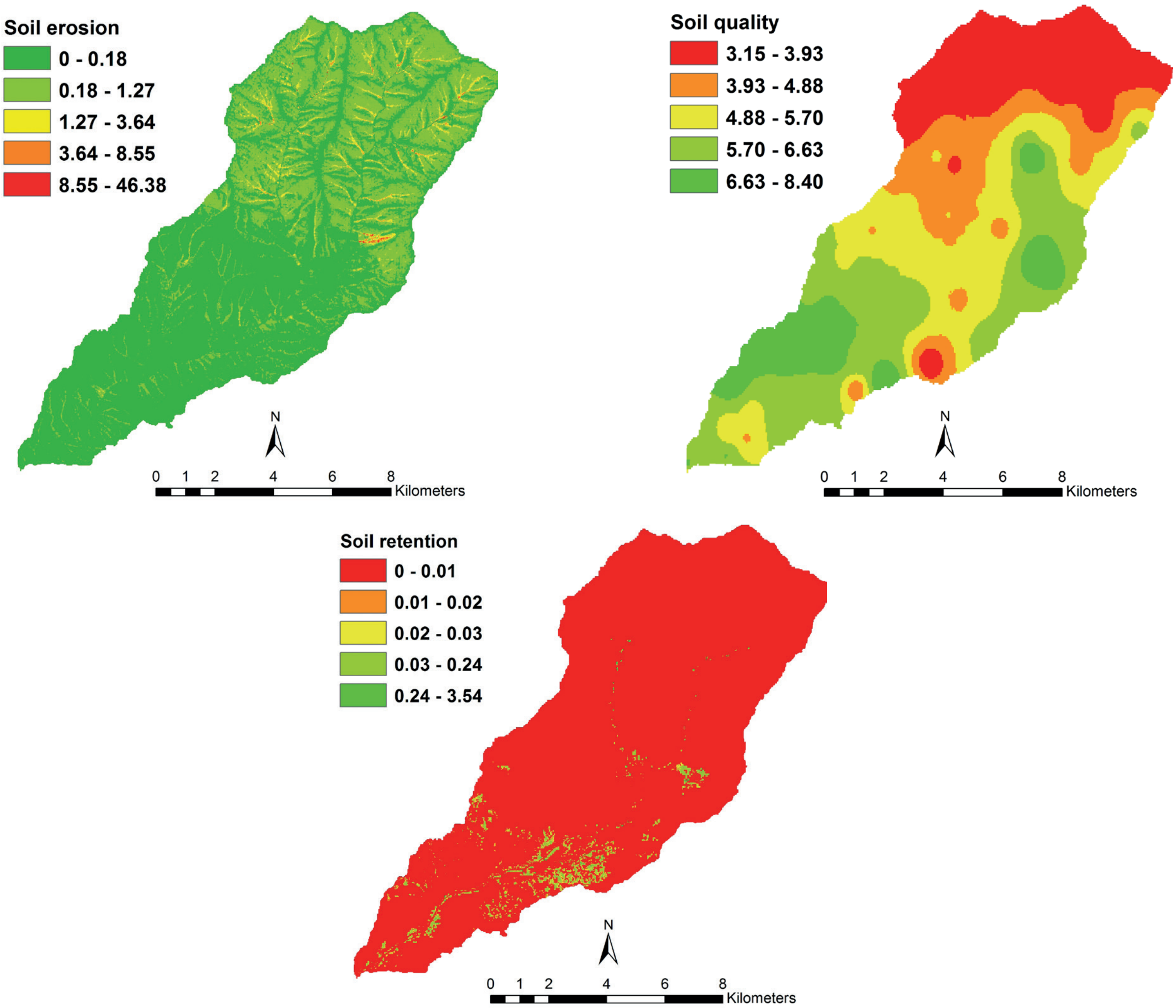

Fig. 5. Maps of average soil erosion (t ha^{-1} yr^{-1}), average soil quality, and average soil retention (t ha^{-1} yr^{-1}) in the study area

Table 4. Correlation between soil quality (SQ), soil retention (SR), and soil erosion (SE)

Index	SQ	SR	SE
SQ	1		
SR	0.86**	1	
SE	-0.45**	-0.35*	1

** p < 0.01, *p < 0.05

loss from hillsides and terraces (up to 55 t ha^{-1} year^{-1}) is greater than that from plain lowlands (up to 3 t ha^{-1} year^{-1}) in northeastern Iran.

Soil erosion decreases from the upstream to downstream sections of the Zar-Abad catchment. The upstream areas of the study catchment are very sensitive to soil erosion due to steep slopes, high rainfall erosivity, high soil erodibility, and low vegetation cover. Downstream areas of the study catchment experience less soil erosion due to improved vegetation cover (irrigated farming) and less steep slopes. Currently, rangelands make up the dominant land use in the upstream parts of the study catchment, which show the highest rates of soil erosion on steep slopes and in areas without vegetation cover. These results are similar to the findings of Derakhshan-Babaei et al. (2021) in the Kan catchment, north of Tehran, Iran. In that study, erosion rates were high in the upstream regions with steep slopes and scarce vegetation. Furthermore, erosion rates in rangelands were higher than those estimated for rainfed and irrigated (orchard) farming lands. Rangelands with poor vegetation cover experience increased erosion rates, while orchards with more vegetation and more organic matter (organic manures are added to the soils) exhibit lower erosion rates. According to Guerra et al. (2016), vegetation cover can regulate soil loss. In the upstream sections of the study area, rainfall quickly turns into surface runoff due to steep gradients, which reduce infiltration. The erosive energy of the resulting flows is increased by the lack of surface cover.

Soil quality in the study area improved from the upstream to the downstream parts of the catchment. This is because, in addition to the soil's physical and chemical properties, the upstream area has a steep slope and ridge topography, whereas the downstream portion has gentler slopes and valley formations. A similar pattern was reported by Derakhshan-Babaei et al. (2021). Further downstream in the Zar-Abad catchment, rainfed and irrigated farming (orchards) are dominant, and here, soil quality is higher. Similar results have been reported by Rahmanipour et al. (2014) and Fang et al. (2024). Furthermore, Ma et al. (2024) demonstrated that cultivation and soil erosion play a significant role in the degradation of soil quality.

Soil quality differed across the three land use types: irrigated farming lands, rangeland, and rainfed farming lands. Soil quality was statistically significantly different between irrigated farming and rangelands. However, there was no significant difference between irrigated and rainfed farming lands. Davari et al. (2020) previously reported that soil quality differs between irrigated farming and dry farmlands. In our study area, soil quality in irrigated farming is higher than in rainfed and rangeland areas, and the former areas have better soil quality than the latter. Higher soil quality in orchards and irrigated farming likely reflects various factors, including crop residues and the addition of inorganic or organic fertilisers. This result contrasts with Nabiollahi et al. (2018), who reported high soil quality in rangelands of western Iran compared to croplands due to various controls on soil erodibility, such as lithology, relief, and vegetation cover.

The soil retention service increased from the upstream to the downstream portion of the study catchment. This area has lower rainfall erosivity, lower soil erodibility, and less steep slopes. Less steep slopes and improved vegetation cover in the downstream areas encourage higher infiltration, thus reducing runoff and erosion. Xiao et al. (2017) previously reported that high vegetation cover is associated with high soil conservation values. The SQI exhibited a strong correlation with soil retention. Soil quality and soil retention exhibited strong negative correlations with soil erosion. Over time, the relationships between soil attributes pertaining to soil quality indicators and regulating services have attracted increasing attention. For example, Van Eekeren et al. (2010) reported a significant relationship between soil physical and biological attributes and the provision of ecosystem services. Black et al. (2010) reported the important role of soil carbon in the delivery of ecosystem services by soils.

Due to erosion, soil nutrients, organic matter, and microorganisms are mobilised and, depending on the hillslope gradient, are deposited on the foot slope or toe slope. This typically improves soil quality in low-altitude land, although it depends on the sediment delivery ratio of the catchment in question. Our work herein did not consider the potential control exerted by the sediment delivery ratio. In addition, where resources permit, soil quality should be investigated for discrete soil horizons by sampling soil at different depths, as erosion and weathering processes can affect the soil properties associated with soil quality (Nosrati and Collins, 2019).

A methodological limitation of our study relates to the use of NDVI for parameterising the C-factor in RUSLE. While NDVI is one of the most widely applied proxies of vegetation cover due to its simplicity and accessibility from multispectral imagery, it is not without shortcomings. Panagos et al. (2015) explicitly noted that NDVI was not adopted in their European-scale soil erosion assessment because of its relatively weak correlation with vegetation attributes. This was partly caused by soil background reflectance and variations in vegetation vitality (de Asis and Omasa, 2007; Vrieling, 2006). These issues may introduce uncertainty in C-factor estimation, particularly in sparsely vegetated or heterogeneous landscapes. Nevertheless, despite these limitations, NDVI-based approaches remain commonly used in RUSLE applications (e.g., Ayalew et al., 2020). This is because they provide spatially explicit estimates of vegetation cover, which are often the only practical option in data-scarce environments. Future studies could benefit from integrating multi-date imagery, soil-adjusted vegetation indices (e.g., SAVI) or field-based measurements to improve the reliability of C-factor estimates.

CONCLUSION

In this study, the (SQI) was applied to represent the effect of soil erosion on soil retention as a regulating ecosystem service. A strong negative correlation was observed between soil erosion, SQI, and soil retention. These results demonstrate that establishing appropriate management systems in an area exposed to soil erosion can contribute to a reduction in soil erosion, the maintenance of soil quality, and an increase in soil retention. Importantly, not all of these responses will occur quickly. This means that soil loss and associated reductions in regulating services pose a serious threat to soil regeneration, global food security, and human survival. Further research into, development of, and validation of SQIs are required to identify and quantify the impact of soil erosion on various essential ecosystem services.

REFERENCES

Afshar F.A., Ayoubi S. and Jalalian A. (2010). Soil redistribution rate and its relationship with soil organic carbon and total nitrogen using 137Cs technique in a cultivated complex hillslope in western Iran. Journal of Environmental Radioactivity, 101, pp.606-614.

Aitkenhead M.J. and Coull M.C. (2019). Digital mapping of soil ecosystem services in Scotland using neural networks and relationship modelling. Part 2: Mapping of soil ecosystem services Soil Use and Management, 35, 217-231.

Amuah E.E.Y., Fei-Baffoe B., Kazapoe R.W., Dankwa P., Okyere I.K., Sackey L.N.A., Nang D.B. and Kpiebaya P. (2024). From the ground up: Unveiling Ghana's soil quality crisis and its ecological and health implications. Innovation and Green Development, 3, 100097.

Andrews S.S. and Carroll C.R. (2001). Designing a soil quality assessment tool for sustainable agroecosystem management. Ecological Applications, 11, pp.1573-1585.

Andrews S.S., Karlen D.L. and Cambardella C.A. (2004). The soil management assessment framework: A quantitative soil quality evaluation method. Soil Science Society of America Journal, 68, pp.1945-1962.

Arshad M.A. and Martin S. (2002). Identifying critical limits for soil quality indicators in agro-ecosystems. Agriculture, Ecosystems & Environment, 88, pp.153-160.

Ayoubi S., Emami N., Ghaffari N., Honarjoo N. and Sahrawat K.L. (2014). Pasture degradation effects on soil quality indicators at different hillslope positions in a semiarid region of western Iran. Environmental Earth Sciences, 71, pp.375-381.

Aziz I., Mahmood T., Raut Y., Lewis W., Islam R. and Weil R. (2009). Active organic matter as a simple measure of field soil quality. ASA International Meetings, Pittsburg, PA.

Black H.J., Glenk K., Lilly A., Artz R., Watson C., Colombo S., Brown I. and Towers W. (2010). What are the opportunities for enhancing ecosystem services from soils through management of soil carbon? In: Gilkes R.J. and Prakongkep N. (eds), Proceedings of the 19th world congress of soil science: Soil solutions for a changing world. International Union of Soil Sciences (Brisbane, Australia, 1–6 August 2010).

Blake G. and Hartge K. (1986). Particle density Methods of soil analysis: Part 1 physical and mineralogical methods, 5, 377-382.

Bo-Jie F., Shi-Liang L., Li-Ding C., Yi-He L. and Jun Q. (2004). Soil quality regime in relation to land cover and slope position across a highly modified slope landscape. Ecological Research, 19, pp.111-118.

Carter M.R. and Gregorich E.G. (2007). Soil Sampling and Methods of Analysis. CRC press,

Comerford N.B., Franzluebbers A.J., Stromberger M.E., Morris L., Markewitz D. and Moore R. (2013). Assessment and evaluation of soil ecosystem services. Soil Horizons, 54(3), pp.1-14.

Corwin D. and Lesch S. (2003). Application of soil electrical conductivity to precision agriculture: theory, principles, and guidelines. Agronomy Journal, 95, pp.455-471.

Davari M., Gholami L., Nabiollahi K., Homaee M. and Jafari H.J. (2020). Deforestation and cultivation of sparse forest impacts on soil quality (case study: West Iran, Baneh). Soil and Tillage Research, 198, 104504.

Derakhshan-Babaei F., Nosrati K., Mirghaed F.A. and Egli M. (2021). The interrelation between landform, land-use, erosion and soil quality in the Kan catchment of the Tehran province, central Iran CATENA, 204, 105412.

Dilly O., Pompili L. and Benedetti A. (2018). Soil micro-biological indicators separated land use practices in contrast to abiotic soil properties at the 50 km scale under summer warm Mediterranean climate in northern Italy. Ecological Indicators, 84, pp.298-303.

Dominati E., Patterson M. and Mackay A. (2010). A framework for classifying and quantifying the natural capital and ecosystem services of soils. Ecological Economics, 69, pp.1858-1868.

Dominati E.J. (2013). Natural capital and ecosystem services of soils. In: Dymond J.R. (ed), Ecosystem Services in New Zealand: Conditions and trends. Manaaki Whenua Press (Lincoln, New Zealand), pp.132-142.

Doran J. and Safley M. (1997). Defining and assessing soil health and sustainable productivity. in: C. Pankhurst, B.M.D.V.V.S.R.G. (Ed.). Biological Indicators of Soil Health, Wallingford CAB INTERNATIONAL pp.1-28.

Doran J.W. and Parkin T.B. (1994). Defining and assessing soil quality. In: J.W. Doran, D.C. Coleman, D.F. Bezdicek and B.A. Stewart, Defining Soil Quality for a Sustainable Environment. s:SSSA Special Publications, pp. 1-21. DOI: https://doi.org/10.2136/sssaspecpub35.c1.

Emadodin I., Narita D. and Bork H.R. (2012). Soil degradation and agricultural sustainability: an overview from Iran. Environment, Development and Sustainability, 14, pp.611-625.

Eskandari Dameneh H., Gholami H., Telfer M.W., Comino J.R., Collins A.L. and Jansen J.D. (2021). Desertification of Iran in the early twenty-first century: assessment using climate and vegetation indices. Scientific Reports, 11, 20548.

Fang H., Zhai Y. and Li C. (2024). Evaluating the impact of soil erosion on soil quality in an agricultural land, northeastern China. Scientific Reports, 14, 15629.

Fisher B., Turner R.K. and Morling P. (2009). Defining and classifying ecosystem services for decision making. Ecological Economics, 68, pp.643-653.

Garrigues E., Corson M.S., Angers D.A., van der Werf H.M. and Walter C. (2013). Development of a soil compaction indicator in life cycle assessment. The International Journal of Life Cycle Assessment, 18, pp.1316-1324.

Gburek W.J., Sharpley A.N., Heathwaite L. and Folmar G.J. (2000). Phosphorus management at the watershed scale: A modification of the phosphorus index. Journal of Environmental Quality, 29, pp.130-144.

Gómez-Baggethun E., de Groot R., Lomas P.L. and Montes C. (2010). The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes. Ecological Economics, 69, pp.1209-1218.

Guerra C.A., Maes J., Geijzendorffer I. and Metzger M.J. (2016). An assessment of soil erosion prevention by vegetation in Mediterranean Europe: Current trends of ecosystem service provision. Ecological Indicators, 60, pp.213-222.

Helmke P.A. and Sparks D.L. (1996). Lithium, sodium, potassium, rubidium, and cesium Methods of soil analysis: Part 3 chemical methods, 5, pp.551-574.

Irvem A., Topaloğlu F. and Uygur V. (2007). Estimating spatial distribution of soil loss over Seyhan River Basin in Turkey. Journal of Hydrology, 336, pp.30-37.

Jalalian A., Ghahareh M. and Karimzadeh H. (1994). Erosion and sediment in some drainage basin of Iran. 4th Iranian Soil sciences Congress, Isfahan, 9-15.

Jong S.M.d. (1994). Applications of reflective remote sensing for land degradation studies in a Mediterranean environment. Nederlandse geografische studies, 177.

Juhos K., Szabó S. and Ladányi M. (2016). Explore the influence of soil quality on crop yield using statistically-derived pedological indicators. Ecological Indicators, 63, pp.366-373.

Karlen D.L., Mausbach M., Doran J.W., Cline R., Harris R. and Schuman G. (1997). Soil quality: a concept, definition, and framework for evaluation. Soil Science Society of America Journal, 61, pp.4-10.

Kroetsch D. and Wang C. (2008). Particle size distribution. In: M.R. Carter E.G. Gregorich, Soil Sampling and Methods of Analysis, pp. 713-725.

Lal R. (2015). Restoring soil quality to mitigate soil degradation. Sustainability, 7, pp.5875-5895.

Landau S. and Everitt B.S. (2003). A handbook of Statistical Analyses Using SPSS. Chapman and Hall/CRC,

Lufafa A., Tenywa M., Isabirye M., Majaliwa M. and Woomer P. (2003). Prediction of soil erosion in a Lake Victoria basin catchment using a GIS-based Universal Soil Loss model. Agricultural Systems, 76, pp.883-894.

Ma R., Tian Z., Zhao Y., Wu Y. and Liang Y. (2024). Response of soil quality degradation to cultivation and soil erosion: A case study in a Mollisol region of Northeast China. Soil and Tillage Research, 242, 106159.

Marzaioli R., d'Ascoli R., De Pascale R. and Rutigliano F. (2010). Soil quality in a Mediterranean area of Southern Italy as related to different land use types. Applied Soil Ecology, 44, pp.205-212.

Mousazadeh R., Badamfirooz J. and Rahmati A. (2018). Scenario modelling to predict future changes in land cover and/or land use, using InVEST software (case study: Dohezar and Sehezar forested landscape). Environmental Sciences, 16, pp.111-132.

Nabiollahi K., Golmohamadi F., Taghizadeh-Mehrjardi R., Kerry R. and Davari M. (2018). Assessing the effects of slope gradient and land use change on soil quality degradation through digital mapping of soil quality indices and soil loss rate. Geoderma, 318, pp.16-28.

Nosrati K. and Collins A.L. (2019) A soil quality index for evaluation of degradation under land use and soil erosion categories in a small mountainous catchment, Iran. Journal of Mountain Science, 16, pp.2577-2590.

Nosrati K. and Van Den Eeckhaut M. (2012). Assessment of groundwater quality using multivariate statistical techniques in Hashtgerd Plain, Iran. Environmental Earth Sciences, 65, pp.331-344.

Obidike-Ugwu E.O., Bege B., Ariwaodo J.O. and Nwafor O.E. (2025). Land-use changes and management impact on soil quality indicators in tropical ecosystem. Environment, Development and Sustainability, 27, pp.3513-3527.

Palmer C., Smith W. and Conkling B. (2002). Development of a protocol for monitoring status and trends in forest soil carbon at a national level. Environmental Pollution, 116, pp.S209-S219.

Paul C., Kuhn K., Steinhoff-Knopp B., Weißhuhn P. and Helming K. (2021). Towards a standardization of soil-related ecosystem service assessments. European Journal of Soil Science, 72, pp.1543-1558.

Phinzi K., Ngetar N.S. and Ebhuoma O. (2021). Soil erosion risk assessment in the Umzintlava catchment (T32E), Eastern Cape, South Africa, using RUSLE and random forest algorithm. South African Geographical Journal, 103, pp.139-162.

Rahmanipour F., Marzaioli R., Bahrami H.A., Fereidouni Z. and Bandarabadi S.R. (2014). Assessment of soil quality indices in agricultural lands of Qazvin Province, Iran. Ecological Indicators, 40, pp.19-26.

Renard K., Yoder D., Lightle D. and Dabney S. (2011). Universal Soil Loss Equation and Revised Universal Soil Loss Equation Handbook of erosion modelling, 137.

Renison D., Hensen I., Suarez R., Cingolani A.M., Marcora P. and Giorgis M.A. (2010). Soil conservation in Polylepis mountain forests of Central Argentina: is livestock reducing our natural capital? Austral Ecology, 35, pp.435-443.

Robbins C. and Wiegand C. (1990). Field and Laboratory Measurements. ASCE Manual and Reports.

Rojas J.M., Prause J., Sanzano G.A., Arce O.E.A. and Sanchez M.C. (2016). Soil quality indicators selection by mixed models and multivariate techniques in deforested areas for agricultural use in NW of Chaco, Argentina. Soil and Tillage Research, 155, pp.250-262.

Şenlikci A., Doğu M., Eren E., Çetinkaya E. and Karadağ S. (2015). Pressure calcimeter as a simple method for measuring the CaCO3 content of soil and comparison with Scheibler calcimeter Toprak Su Dergisi.

Sharp R., Tallis H., Ricketts T., Guerry A., Wood S.A., Chaplin-Kramer R., Nelson E., Ennaanay D., Wolny S. and Olwero N. (2014). InVEST user's guide The Natural Capital Project, Stanford.

Steinhoff-Knopp B., Kuhn T.K. and Burkhard B. (2021). The impact of soil erosion on soil-related ecosystem services: development and testing a scenario-based assessment approach Environmental Monitoring and Assessment, 193, pp.1-18.

Sun W., Shao Q., Liu J. and Zhai J. (2014). Assessing the effects of land use and topography on soil erosion on the Loess Plateau in China. Catena, 121, pp.151-163.

Van Eekeren N., De Boer H., Hanegraaf M., Bokhorst J., Nierop D., Bloem J., Schouten T., De Goede R. and Brussaard L. (2010). Ecosystem services in grassland associated with biotic and abiotic soil parameters. Soil Biology and Biochemistry, 42, pp.1491-1504.

Van Leeuwen W.J. and Sammons G. (2004). Vegetation dynamics and erosion modeling using remotely sensed data (MODIS) and GIS. USDA Forest Service Tenth Biennial Remote Sensing Applications Conference, 5-9 April.

Walkley A. and Black I.A. (1934). An examination of the Degtjareff method for determining soil organic matter and a proposed modification of the chromic acid titration method. Soil Science, 37, 29-38.

Wang P., Nair M.S., Liu L., Iketani S., Luo Y., Guo Y., Wang M., Yu J., Zhang B. and Kwong P.D. (2021). Antibody resistance of SARS-CoV-2 variants B. 1.351 and B. 1.1. 7. Nature, 593, 1pp.30-135.

Wijesundara N., Abeysingha N. and Dissanayake D. (2018). GIS-based soil loss estimation using RUSLE model: A case of Kirindi Oya river basin, Sri Lanka. Modeling Earth Systems and Environment, 4, pp.251-262.

Wischmeier W.H. (1978). Predicting rainfall erosion losses: a guide to conversation planning USDA Agr. Handbook.

Wischmeier W.H. and Smith D.D. (1978). Predicting rainfall erosion losses: a guide to conservation planning. Department of Agriculture, Science and Education Administration.

Xiao Q., Hu D. and Xiao Y. (2017). Assessing changes in soil conservation ecosystem services and causal factors in the Three Gorges Reservoir region of China. Journal of Cleaner Production, 163, pp.S172-S180.

Zhang K., Yu Y., Dong J., Yang Q. and Xu X. (2019). Adapting & testing use of USLE K factor for agricultural soils in China. Agriculture, Ecosystems & Environment, 269, pp.148-155.

Zhang Y.-G., Nearing M., Zhang X.-C., Xie Y. and Wei H. (2010). Projected rainfall erosivity changes under climate change from multimodel and multiscenario projections in Northeast China. Journal of Hydrology, 384, pp.97-106.