Land-use change from arable lands to orchards reduced soil erosion and increased nutrient losses in a small catchment

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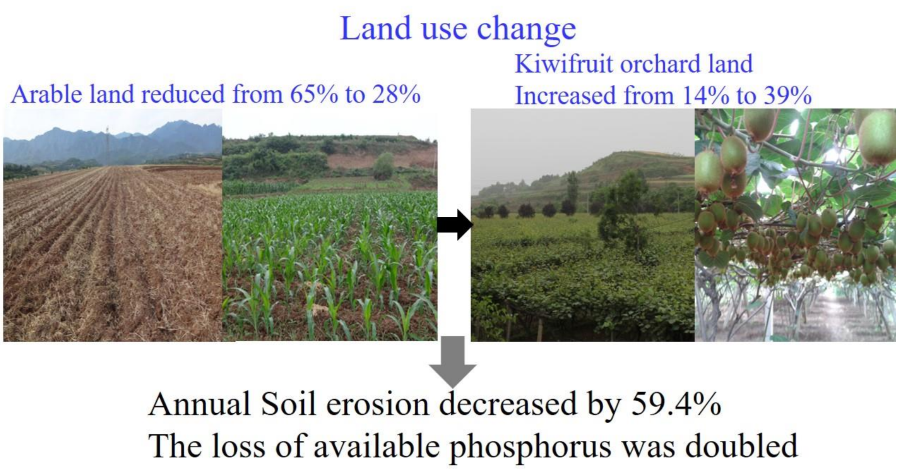
HIGHLIGHTS

- RUSLE and GIS methods were used to evaluate soil erosion and nutrient losses
- Land-use in the catchment was changed from crop to kiwifruit orchards since 1990
- The orchards significantly reduced soil erosion
- More available phosphorus was lost from the orchards

GRAPHICALABSTRACT



Abstract: China has been facing rapid land-use changes since 1980s. For example, more and more arable lands have been converted to orchards in China's Loess Plateau (640,000 km^2) to produce high-value fruits, such as kiwifruits and apples. Over-fertilisation is very common in China's orchards. It is urgent and timely to assess the effects of land-use change on soil erosion and nutrient losses to support sustainable land-use and environmental management in China. The case study area Yujia River Catchment, where many arable lands growing wheat and maize (1957–1989) have been converted to kiwifruit orchards (1990–2013), is a typical hilly region at the northern foot of the Qinling Mountains in Shaanxi. The Revised Universal Soil Loss Equation (RUSLE) model, geographical information systems (GIS) and Remote Sensing were adopted to evaluate the effects of land-use change on soil erosion and nutrient losses in the catchment. The results show that the total soil erosion in the Yujia River Catchment during 1957–2013 was 10.90×10^5 t, equivalent to 0.36 cm thick topsoil lost annually. This is in line with the sediments in the reservoir at the catchment outlet. In the study area, arable land is the major source of soil erosion and its erosion modulus is 9.6 times that of the orchards. The historical land-use change from arable land to orchards since 1990 has reduced the soil-erosion intensity from a strong level to the medium one. The arable lands covering 28% of the catchment have contributed 81.3% of the total loss of organic matter and 80.4% of the total nitrogen loss respectively. However, the loss of available phosphorus mainly occurred in the orchards, representing 66.7% of the total available phosphorus loss in the catchment. The soil-erosion modulus of the arable land is highly sensitive to the land slope. The results of this study are useful in planning the land-use change and in finding measures to reduce the risks of soil erosion and non-point-source pollution in the catchment, such as changing sloping lands to terraces, contour cropping, and comprehensive nutrient management.

Keywords: soil erosion; RUSLE; GIS; land-use change; soil nutrient loss

1. Introduction

The increasingly limited land resource is vital for human survival. However, it is threatened by soil erosion, which causes both loss of the fertile topsoil and reduction of the soil quality. Moreover, the losses of nitrogen and phosphorus due to soil erosion can lead to agricultural non-point-source water pollution (Di Stefano *et al.*, 2016; Hancock *et al.*, 2015;

Tanyas *et al.*,2015) and hence environmental problems e.g. eutrophication. As a whole, Earth's land masses are experiencing considerable erosive soil losses. China is one of the countries suffering the most serious soil erosion in the world (Wang *et al.*, 2012; MWR and NBS, 2013). Approximately 37.6% of the land in China is degraded by soil erosion (Gao *et al.*, 2016; Zhang *et al.*, 2012). For example, the Loess Plateau (640,000 km^2) in northwest China, which is among the world's highest soil erosion rates, generates about 1.6 billion tons of sediment clogging the Yellow River annually and posing a serious downstream flood risk (Gao *et al.*, 2016; Zhang *et al.*, 2012).

The intensity of soil erosion and non-point-source water pollution are closely linked to the land-uses (Ferreiraa *et al.*,2015; Häring *et al.*, 2014; Leh *et al.*, 2013; Li *et al.*, 2016). Many studies have shown that land-use or vegetation change have a big impact on soil erosion (Diyabalanage *et al.*, 2017; Ganasri *et al.*, 2015; Zhang *et al.*, 2017). The structure of China's agriculture has substantially changed in recent years. In particular, driven by the economic development and life-quality improvements in China, large areas of conventional cereal production have been converted to crops with a high economic value, such as kiwifruit and apple trees. (Qiu *et al.*, 2010; Wei *et al.*, 2015; Yu *et al.*, 2014). Consequently, the overall agricultural activities have been dramatically changed in these regions. For example, fruit trees are associated with excessive nutrient and irrigation application in comparison with cereal crops. Meanwhile, over-fertilisation is very common in orchards with intensive management due to limited scientific knowledge and production-management expertise of the farmers in China (Cai *et al.*, 2014; Ju *et al.*, 2014; Zhu *et al.*, 2013). Accordingly, China's policy makers urgently need to understand how this land-use-change trend affects soil erosion and non-point-source water pollution. Therefore, it is timely to investigate the impacts of the land-use change on soil erosion and nutrient loss on sloping fields to better support sustainable management of water and soil conservation and agricultural non-point-source water pollution in China (Ferreiraa *et al.*, 2015; Pang *et al.*, 2012).

The Yujia River Catchment, which is located at the southern edge of the Loess Plateau, is an ideal area to undertake this type of research. This catchment is a typical hilly area at the northern foot of the Qinling Mountains in Shaanxi Province. A large proportion of agricultural land in this catchment has been converted to kiwifruit orchards from arable land planting cereals since 1990 (Wei *et al.*, 2015). This area has become the most important kiwifruit production base in China, accounting for 30% of the total kiwifruit cultivation area in the world and 60% of that in China (Sun and Fu, 2009). The study area is facing the problems of soil erosion and nutrient losses due to high nutrient input in orchards (Lu *et al.*, 2016a, 2016b), high slope of the arable land, Loess soil cemented loosely, and concentrated rainfall between July and September (Wei *et al.*, 2015). Therefore, there is a critical need to undertake this study to provide scientific supports to the local governments to make sound policies for sustainable management of agricultural land resources, soil erosion and water resources.

Numerical modelling plays an important role in investigating soil erosion. It is a common practice to adopt existing soil-erosion models with varying degrees of complexity to understand and estimate soil erosion, using field data and derived data from remote sensing and Geographic Information Systems (GIS) (Fernandez *et al.*, 2003; Xu *et al.*, 2009). The RUSLE model (Renard *et al.* 1997), which has been extensively tested in many soil erosion studies (e.g., Meusburger *et al.* 2010; Rulli *et al.* 2013; Xu *et al.* 2013), is useful in estimating soil erosion on an annual basis due to its parsimonious parameter requirement. In addition, parameters required by RUSLE are readily available or can be easily derived from other datasets. For example, slope can be calculated from digital elevation model (DEM) using GIS; and the land-use map can be derived from satellite images. In this study, RUSLE was adopted in conjunction with GIS and Remote Sensing to estimate the soil erosion and soil nutrient losses in 1957–1989 and 1990–2013 reflecting the typical agricultural land-use changes in the catchment. Our hypothesis was that land-use change from arable lands to kiwifruit orchards would reduce soil erosion and increase nutrient losses in the small catchment.

2. Materials and Methods

2.1. Study site

The Yujia River Catchment (107°39′–108°37′E and 33°42′–34°14′N) is located in the Zhouzhi county, Shaanxi province (Fig. 1), covering an area of 4.12 km^2. Lying at the northern foot of the Qinling Mountains, it is characterised by a V-shaped geomorphologic ravine with an altitude of 487–672 m (Fig. 1). In the catchment, the land surface tips northeastwards and has the lowest terrain located at the outlet of the catchment, where a reservoir was built in 1957 (Fig. 1).

In the catchment, 85% of the arable lands have a slope range of 0–15°. With an annual mean temperature of 13.2°C, the area has an annually average rainfall of 713 mm (1957–2013), of which 61–84% occurred between July and September, indicating a typical temperate continental monsoon climate. The major soil types in the catchment can be divided into three soil groups，i.e. Hapli-Ustic Argosols, Typic Usti-Alluvic Primosols and Loessi-Orthic Primosols (Chinese soil taxonomic classification), and five soil subgroups (Chinese Soil Taxonomy Research Group, 2001).

Before 1989, the arable land, orchard, forest and unutilised land accounted for 65%, 14%, 3.2% and 0.5% of the catchment respectively. However, their proportions were changed to 28%, 39%, 7.1% and 1.5% from 1989 when China’s agricultural structure adjustments began. The main cereals in the catchment are winter wheat and summer corn, while kiwifruit is the leading fruit (occupying 84% of the orchards) due to its high economic value.

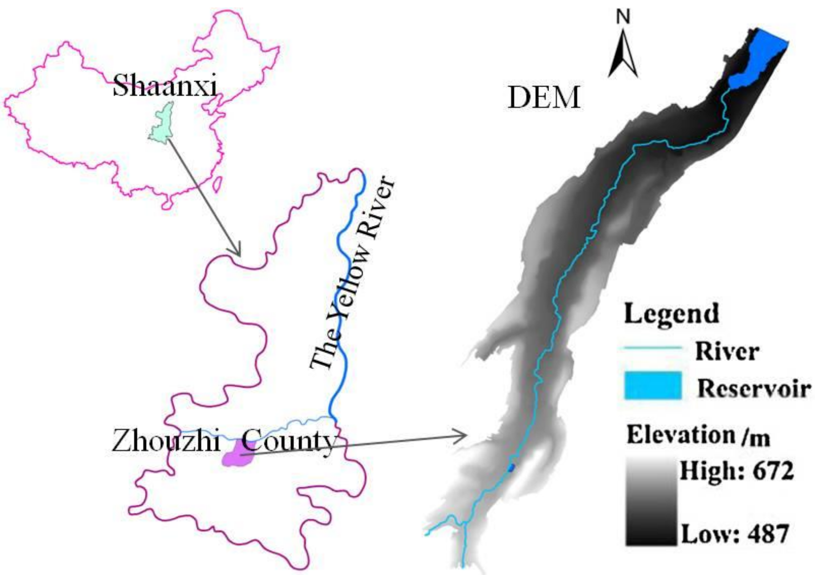


Fig. 1 The location and digital elevation model (DEM) of the Yujia River Catchment, Shaanxi province, China.

2.2. The Revised Universal Soil Loss Equation (RUSLE) model

The RUSLE model was developed to predict long-term average annual soil loss (t ha^{-1} year^{-1})

resulting from raindrop splash and runoff from specific field slopes in specified cropping and management systems (Wischmeier and Smith, 1978). RUSLE has been widely used to generate reliable estimate of soil erosion (Borrelli *et al.*, 2017; Di Stefano *et al.*, 2016; Haregeweyn *et al.*, 2017; Leh *et al.*, 2013; Panagos *et al.*, 2015; SooHoo *et al.*, 2017; Tanyas *et al.*, 2015). It considers five factors affecting erosion, namely rainfall-runoff erosivity (R), (MJ mm ha^{-1} h^{-1} $year^{-1}$), soil erodibility (K) (t ha^{-1} h ha^{-1} MJ^{-1} mm^{-1}), topographic factor (the combination of the slope length and the steepness factor) (LS), crop management factor (C), and the factor of erosion-control practice (P). The RUSLE equation can be written as follow:

$$A = R \times K \times LS \times C \times P \qquad (1)$$

where A is soil annual loss (t ha^{-1} $year^{-1}$).

2.2.1. *R*–rainfall-runoff erosivity factor

R is the major driving force of the soil erosion, reflecting the effect of raindrop impact and the amount and rate of runoff likely to be associated with rain. In the original USLE method, R was defined as a function of kinetic energy and 30-minute rainfall intensity derived from measurements of autographic recorders (Wischmeier and Smith, 1978). Since the time series of the rainfall intensity are not readily available, the method of Renard (Haregeweyn *et al.*, 2013; Leh*et al.*, 2013; Renard *et al.*, 1997) was adopted in this study to calculate *R* using annual rainfall in 57 years (1957–2013):

$$R = \begin{cases} 0.0483Pa^{1.61} & Pa \le 850\,mm \\ 587.8 - 1.249Pa + 0.004105Pa^2 & Pa > 850\,mm \end{cases} \qquad (2)$$

where, *Pa* is the annual rainfall (mm).

2.2.2. *K*–Soil erodibility factor

K values, which represent the effect of soil properties and soil profile characteristics on soil loss, reflect the rate of soil loss per rainfall-runoff erosivity (*R*). Normally *K* is estimated based on soil textures and organic matter. *K* can be calculated using equation below (Williams, 1990):

$$K = \{0.2 + 0.3\exp[-0.0256SAN(1-\frac{SIL}{100})]\} \cdot (\frac{SIL}{CLA+SIL})^{0.3} \cdot [1.0 - \frac{0.25SOC}{SOC+\exp(3.72-2.95SOC)}] \cdot [1.0 - \frac{0.7SN_1}{SN_1+\exp(-5.51+22.9SN_1)}] \quad (3)$$

where, SAN, SIL and CLA are the mass fractions (%) of sand, silt and clay respectively; $SN_1 = 1 - SAN/100$; and SOC is the mass fraction of soil organic carbon (%). The soil types (Fig. 2), SAN, SIL, CLA and SOC were extracted from the soil database of the second national soil census of China.

The value of K in the formula above is in the American unit of t·acre·hr·100^{-1}·$acre^{-1}$·$feet^{-1}$·ton^{-1}·$inch^{-1}$ (Williams, 1990). Therefore, a constant value of 0.1317 was used to transform K into the international unit of t·ha·hr·ha^{-1}·MJ^{-1}·mm^{-1}.

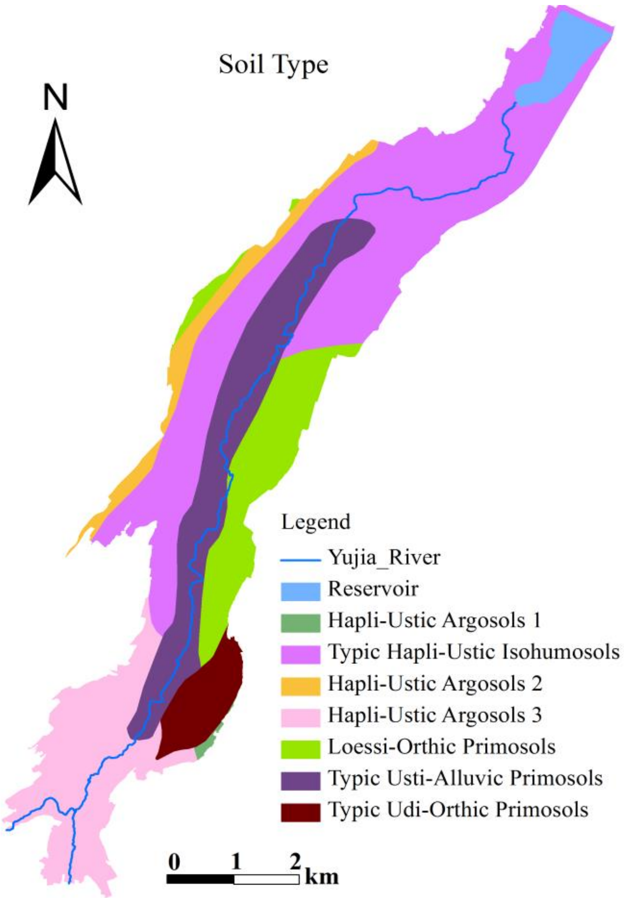


Fig. 2 The distribution of soil type in the Yujia River Catchment, Shaanxi province, China. Hapli-Ustic Argosols 1, Hapli-Ustic Argosols 2, Hapli-Ustic Argosols 3 are three unnamed subgroups in the soil family of Hapli-Ustic according to Chinese soil taxonomic classification.

2.2.3. *LS*–Topographic factor

LS represents the ratio of soil loss on a given slope length and steepness to soil loss from a slope that has a length of 22.1 m and a steepness of 9% (Renard *et al.*, 1997). The slope length (L) and slope steepness (S) were calculated using DEM (1m ×1m) in the study area (Fig. 1, Table 1). The method of Wischmeier and Smith (1978) was adopted to calculate L:

$$L = (\lambda / 22.1)^{\alpha}$$

$$\alpha = \beta / (1+\beta) \qquad (4)$$

$$\beta = (\sin\theta / 0.089) / [3.0(\sin\theta)^{0.8} + 0.56]$$

where, λ is the horizontal projection of the slope length; α is the index of the slope length calculated using the method of Renard *et al.* (1997); and θ is the slope steepness. Since there is a wide variation in slope in the study area, the McCool Formula (McCool *et al.*, 1987) was used to calculate S at the locations with a gentle slope ($\theta \leq 5°$):

$$S = 10.8\, sin\,\theta + 0.03 \qquad (5)$$

The method of liu *et al.* (2001) was used for S calculation where $\theta > 5°$:

$$S = \begin{cases} 16.8\sin\theta - 0.5 & 5° \leq \theta < 10° \\ 21.9\sin\theta - 0.96 & \theta \geq 10° \end{cases} \qquad (6)$$

2.2.4. *C*–Crop management factor

C, which reflects the effect of cropping and management activities on erosion rates, indicates how the conservation plan will affect the average annual soil loss. The *C* value ranges from zero to one. A zero value indicates that soil erosion is unlikely to occur; and a value of one suggests unrestrained soil erosion. *C* values in this study were from the study of Cai *et al.* (2000) undertaken based on the land-use map (Table 1, Fig. 3). Arable land, orchard, forest, residential land, roads, waters, and unutilised land were assigned the *C* values of 0.31, 0.05, 0.006, 0, 0, 0 and 0.4 respectively. Based on the history of agricultural structural adjustments, year 1990 was treated as the key turning point of the land-use change in the area. Fig. 3 shows the distribution of *C* values in the periods of 1957–1989 and 1990–2013 derived from the land-use patterns of 1989 and 2013. The land-use map of 2013, which was derived by interpreting the Quickbird multispectral satellite image, has been validated via the field survey in the study area (Table 1).

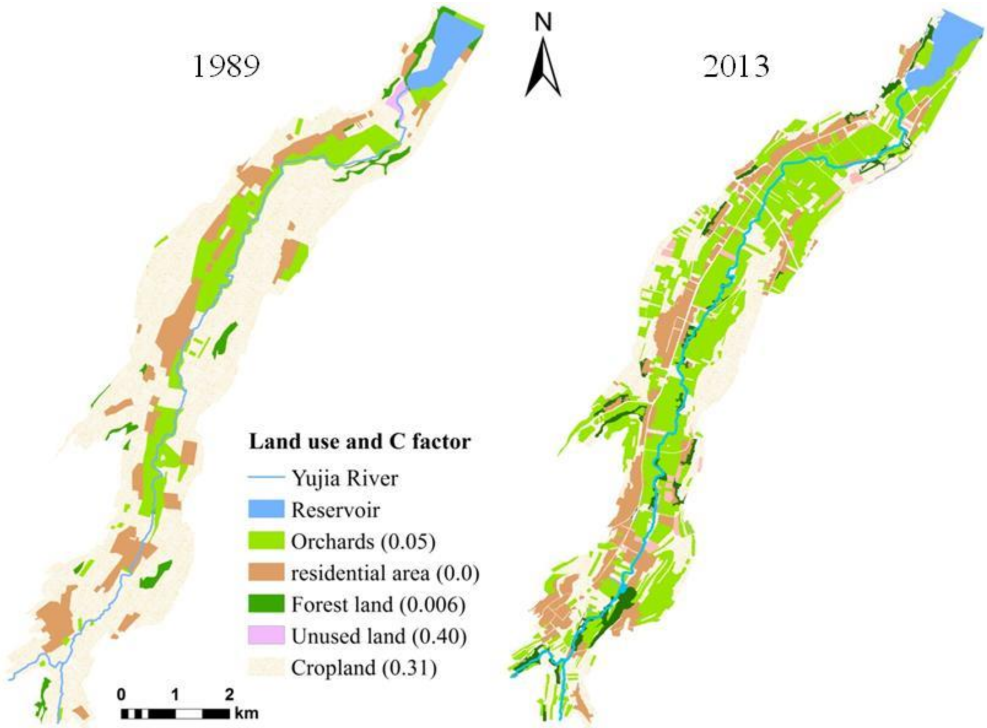


Fig. 3 Land-use maps of 1989 and 2013 in the Yujia River Catchment.

2.2.5. *P*–Conservation practice factor

P represents the ratio of soil loss after the implementation of a conservation practice trying to reduce soil loss from straight-row farming up and down slope. The *P* value ranges from 0 to 1 (P = 0: no soil erosion due to the conservation practices; P = 1: the occurrence of soil erosion at its highest potential rate when lacking conservation practices). Since there were very limited measures of water and soil conservation in the area before 1990, *P* from 1957 to 1989 was assigned a value 1. However, contour farming methods have been applied in most arable lands in the study catchment since 1990. Therefore, P was assigned different values during the period 1990–2013 based on the slope of land: P = 0.3 in areas where $\theta < 5°$; P = 0.5 where $5° \leq \theta < 10°$; and P = 0.6 where $\theta \geq 10°$.

2.3. Datasets

The datasets used for estimating soil erosion in this study are listed in Table 1.

Table 1. The datasets used to calculate the soil erosion

Data type	Source	Description

Digital Elevation Model (DEM)	1:10,000 topographic map, Shaanxi Bureau of Geographic Surveying and Mapping.	We used ArcGIS10.0 Spatial Analyst tool to discretise the cells ca. 1m ×1m in size on the basis of 1:10,000 scale topographic map
Land-uses	1989: Office of Land Resource Investigation of Shaanxi Province	1:10,000 scale
	2013: Interpretation of satellite images in April 2012	Interpretation after the image treatment on the purchased Quickbird multispectral satellite image (0.61m resolution) of April 2012; field survey in May 2013
Soil type data	2nd soil investigation data, sampling analysis of Shaanxi Province	The distribution of soil types and soil textures from the 2nd soil investigation data of Shaanxi Province
Rainfall data	Zhouzhi Meteorological Bureau	Meteorological station near the study area during 1957–2013

2.4. Calculating the dry weight of sediments in the reservoir

The Reservoir at the outlet of the catchment was built in 1957. With a dam of 17m high and 318m long, its initial volume was 11.75×10^5 m^3 when it was built. No dredging work has been undertaken since 1957. As a result of leakage, the reservoir has a low water level during the summer and autumn, and remains dry the rest of the year. We calculated the amount of the sediments deposited in the reservoir by comparing the difference between the reservoir's initial volume and the volume measured in December 2013. The dry density of the sediments in the reservoir was measured as 1.51 g cm^{-3}. The dry weight of sediments in the reservoir over the 57-year period was calculated by multiplying the volume of the sediments by the dry density of sediments.

2.5. Estimating soil nutrient losses from the catchment

The soil nutrient contents in the catchment were measured in October 2012. The grid sampling method (60m×60m) was used to take soil samples within the catchment; and the multi-point composite samples were collected from 118 sampling sites shown in Fig. 4. The location of the central point of each sampling site was recorded using portable GPS device. For each sampling site, the content of soil organic matter, total nitrogen (TN) and available phosphorus (AP) within 0–20 cm soil depth were analysed using the method of Bao (2000). The spatial distribution of the soil contents of organic matter, TN and AP was then generated using the Kriging interpolation method via ArcGIS10.1 (Fig. 4). Based on the average soil nutrient contents, the amount of soil nutrient losses were calculated using the estimated mass of the soil erosion.

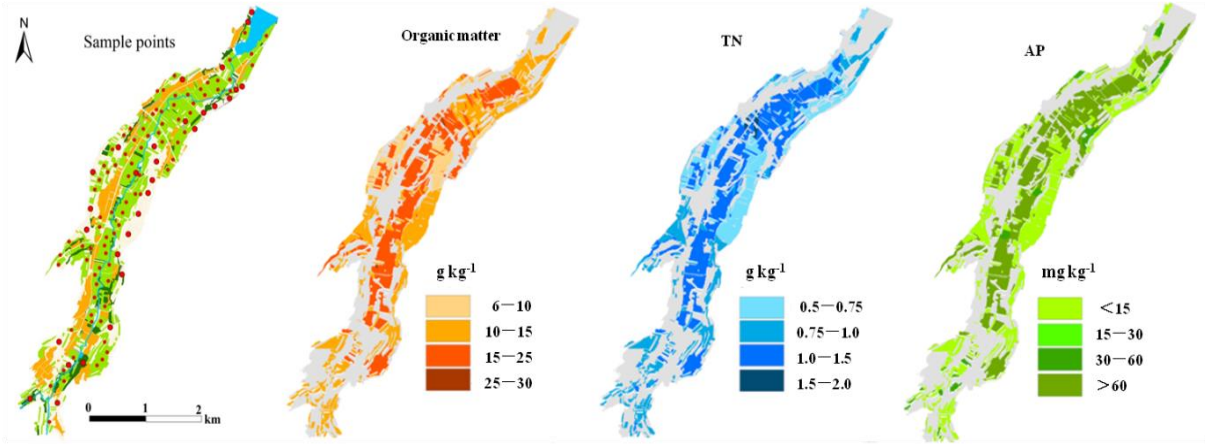


Fig.4 Soil sampling points and the distribution of soil nutrients in the Yujia River Catchment. TN: total nitrogen; AP: available phosphorus.

3. Results

3.1. Verifying the results of the RUSLE method

The total soil erosion estimated using the RUSLE model in the Yujia River Catchment over 57 years is 10.90×10^5 t, based on the different land-use patterns in 1957–1989 and 1990–2013. This estimate is equivalent to a loss of 0.36 cm thick topsoil in the catchment every year. It worth noting that RUSLE was designed to estimate soil erosion instead of sediment yield, which is the erosion from slopes, channels, and mass wasting, minus the sediment deposited after it is eroded but before it reaches the point of interest (Renard *et al.*, 1997). The sediments deposited in the reservoir were measured as 10.11×10^5 t, which is 7.8% less than the total soil erosion estimated using RUSLE. This difference may be caused by the settlement of some of the soil eroded along the streams or rivers in the catchment before reaching the reservoir. Therefore, this indicates that the estimate of soil erosion using RUSLE is sensible in this study.

3.2. Soil erosion intensity

The estimated annual soil erosion modulus in the catchment was categorised into the slight, light, medium, strong, very strong or severe soil erosion intensity, in accordance with the SL190-2007 Standards for Classification and Gradation of Soil Erosion (Ministry of Water Resources of China, 2008). The distributions of soil erosion intensity in the periods of 1957–1989 and 1990–2013 are shown in Fig.5, while their details are listed in Table 2. It indicates that the annual mean soil erosion modulus was 6,185 t/km^2 in 1957–1989, and categorised as strong soil erosion. In contrast, the annual mean soil erosion modulus was 2,510 t/km^2 in 1990–2013, categorised as medium soil erosion. The annual mean soil erosion amount in 1957–1989 was 25,497 t, which is 2.5 times that in 1990–2013 (10,349 t). The lands with strong soil erosion intensity and above in 1957–1989 and 1990–2013 accounted for 33.4% and 13.9% of the study area respectively (Fig. 5); and the soil erosion they generated each year in two periods were 23,017 t and 8,022 t, contributing 90.3% and 77.5% of the annual total soil erosion of the catchment respectively.

Table 2. Soil erosion intensity classification of the Yujia River Catchment.

Soil erosion intensity	**Soil erosion modulus (t·km^{-2}yr^{-1})**	**Area percentage (%)**		**Erosion rate (t·yr^{-1})**	
		1957–1989	**1990–2013**	**1957–1989**	**1990–2013**
Slight	⩽ 500	40.5	59.7	127	195
Light	500 –2 500	14.5	18.2	586	942
Medium	2 500 – 5 000	11.6	8.2	1767	1190
Strong	5 000 – 8 000	6.2	2.6	1515	724
Very strong	8 000 – 15 000	11.2	6.5	5361	2929
Severe	> 15 000	16.0	4.8	16141	4369

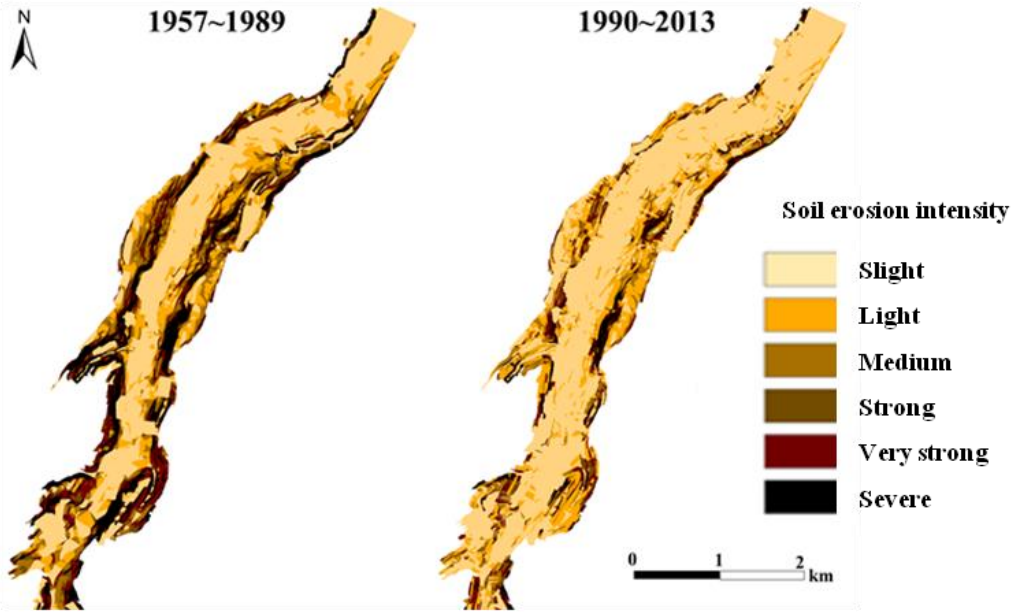


Fig. 5 Annual soil erosion intensity in 1957–1989 and 1990–2013 in the Yujia River Catchment.

3.3. Soil erosion from different land-uses

Based on the land-use map of 2013, it was found that the soil erosion modulus of the arable land was 9.6 and 29.9 times that of orchard and forest respectively (Table 3). Although the arable land covered only 28% of the total area of the catchment, it contributed 83% of the total soil erosion of 2013 in the study area. Whilst orchards covered 39% of the total area of the catchment, they generated only 12% of the total erosion of the catchment in 2013. Meanwhile, forest contributed only 0.69% of the total soil erosion of 2013 in the catchment. This shows that the arable land is a major source of soil erosion in the Yujia River Catchment.

Table 3. Soil erosion amount under different land-uses in the Yujia River Catchment.

Land-use	**Soil erosion modulus ($t \cdot km^{-2} \cdot yr^{-1}$)**	**Area (km^2)**	**Erosion rate ($t \cdot yr^{-1}$)**
Arable land	9834	1.17	11505
Orchard	1025	1.60	1640
Forest	329	0.29	95

3.4. Sensitivity analysis of soil erosion modulus to the slope of land

The sensitivity analysis of erosion modulus to the slope and land-use was undertaken by changing slope stepwise with an interval of 5° for different land-uses of 2013 (Table 4). It shows that the soil erosion modulus of both the arable land and orchard increases with the rises of the slope of land, whereas that of forest becomes stable when slope is higher than 20°. The average increase of the soil erosion modulus for the arable land was 6.1 and 40.8 times that of the orchard land and forest respectively when increasing the slope in this sensitivity analysis. This indicates that the soil erosion in the arable land is much more sensitive to the slope of land than that in orchard and forest.

Table 4.The sensitivity of the soil erosion modulus to slope in different land-uses

Land-use type	**Soil erosion modulus ($t \cdot km^{-2} \cdot yr^{-1}$)**				
	5°	**10°**	**15°**	**20°**	**25°**
Arable land	1117.97	5134.60	11409.70	17330.50	22964.27
Orchard	168.95	790.17	1925.09	2788.56	3762.86
Forest	61.20	286.19	437.31	604.30	596.30

3.5. Soil nutrient losses in the catchment

The soil nutrient losses from different land-uses of the catchment in 2013 have been calculated using the method described in the section 2.5. Although the arable land covered 28% of the study area, it contributed to 81.3% and 80.4% of the catchment's total organic matter and total nitrogen (TN) lost via soil erosion in 2013 (Table 5). However, the loss of the available phosphorus (AP) from soils was mainly from orchards; and 66.7% of the total loss of the AP was from orchards covering 39% of the catchment.

Table 5.Nutrient loss under different land-use patterns in the Yujia River Catchment

Land-use type	Organic matter ($t \cdot yr^{-1}$)	Total Nitrogen ($t \cdot yr^{-1}$)	Available Phosphorus ($t \cdot yr^{-1}$)
Arable land	109.96	7.21	0.06
Orchard land	23.39	1.62	0.12
Forest land	1.91	0.14	-

4. Discussions

4.1. The reduction of soil erosion due to the development of orchards

Compared with the period of 1957–1989, the soil erosion modulus and erosion rate in the catchment during 1990–2013 have decreased significantly (Table 3 and Fig. 5). This was mainly caused by converting the arable land to kiwifruit orchards. The proportion of the arable land in the catchment has reduced from 65% (1957–1989) to 28% (1990–2013), while that of the kiwifruit orchards has increased from 14% (1957–1989) to 39% (1990–2013). The kiwifruit orchards have less soil erosion than the arable land due to the two reasons: (1) The crop rotation is winter wheat and summer maize in the catchment. Wheat is normally harvested in early June, and maize is then sowed immediately. The rainfall starts to increase from June in the study catchment. Therefore, the possibility of the occurrence of soil erosion becomes high during this transition period and early stage of maize (Vogel *et al.*, 2016) due to the bare soil or less vegetation coverage (Vanwalleghem *et al.*, 2017). (2) Irrigation is required in the kiwifruit orchards due to high water demand of kiwifruit vines. In actual practice, kiwifruit growers usually level the land or make small ridges to prevent water quickly running out their kiwifruit orchards, thus leading to the reduction of soil erosion.

4.2. The impacts of land-use change on soil erosion and nutrient losses

The results show that the soil erosion and the loss of the organic matter and TN mainly originated from the arable land, whereas the loss of the soil AP was mainly from orchards in the catchment (Table 5). Compared with available nutrients in soil, the organic matter and TN of soil are more stable and subject to fertilisation and field management. For example, the average contents of the organic matter for the arable land and orchards in the catchment are 11.28 g kg^{-1} and 17.15 g kg^{-1}; while the average contents of TN in two land-uses are 0.74 g kg^{-1} and 1.18 g kg^{-1}, respectively. However, the average content of the soil AP in the kiwifruit orchards (85.67 mg kg^{-1}) is more than 13 times that of the arable land (6.15 mg kg^{-1}) (Wei *et al.*, 2015). That explains why a high proportion of the soil AP was mainly from the kiwifruit orchards in the catchment.

The fertilisation surveys in the catchment showed that phosphate fertiliser has been overused in 51.7% of the kiwifruit orchards; while 81.8% of the kiwifruit orchards have the problem of excessive application of nitrogen fertiliser (Kang *et al.*, 2014; Lu*et al.*, 2016a, 2016b). Consequently, the soils of many kiwifruit orchards in the catchment have high contents of available N and P, thus increasing the risk of water pollution from agricultural non-point sources. Therefore, although the development of kiwifruit orchards in the catchment has reduced the risk of soil erosion, it has increased the losses of available nitrogen and phosphorus from soils.

4.3. Measures to reducing soil erosion and nutrient losses

The results of this study show that soil erosion and losses of soil organic matter and TN mainly occurred in the sloping arable land. It is also found that the soil erosion modulus of the arable land is highly sensitive to the slope of land. Therefore, measures, such as land levelling, changing sloping land to terraces, and conservation farming (including contour planting and straw mulching), should be taken to reduce the soil erosion and hence the losses of the soil organic matter and TN from arable land (Mhazo *et al.*, 2016; Prosdocimi *et al.*, 2016; Vanwalleghem *et al.*, 2017).

As discussed above, the conversion of the arable land to kiwifruit orchards has increased the losses of the soil phosphorus and nitrate in the catchment. For example, Lu *et al.* (2016a) reported that nitrate accumulation in kiwifruit orchard was as high as 793 kg N/ha in 0-200 cm soil profiles, thus leading to high nitrate leaching (Gao *et al.*, 2016). Therefore, attention should be paid to promoting appropriate fertilisation practices and strengthening the comprehensive management of nutrients in the orchards to reduce nutrient losses.

5. Conclusions

The total soil erosion during 1957–2013 estimated using the RUSLE method is in line with the amount of sediment accumulated in the reservoir at the outlet of the Yujiahe River Catchment. The soil eroded in the catchment is equivalent to an annual loss of 0.36 cm thick topsoil. The arable land is the main source of soil erosion; and its erosion modulus is 9.6 times that of the orchards in the catchment. Land-use change from the arable land to the kiwifruit orchards in the catchment since 1990 has significantly reduced soil erosion intensity from the strong to a medium level. In addition, the soil erosion modulus of the arable land has the highest sensitivity to the slope of land. Therefore, different conservation measures, such as changing sloping land to terraces and straw mulching, should be widely implemented in the

Yujiahe River Catchment to reduce soil erosion. About 81.3% of the organic matter loss and 80.4% of the total nitrogen loss in the catchment were from the arable land. In contrast, 66.7% of the loss of total available phosphorus in the catchment was from the kiwifruit orchards. Therefore, it is necessary to promote appropriate fertilisation practices and strengthen the comprehensive nutrient management in orchards to prevent non-point-source water pollution.

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