Identifying the effects of land use change on sediment export: integrating sediment source and sediment delivery in the Qiantang River Basin, China

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Abstract

Dramatic land use change caused by the rapid economic development in China has impacted the sediment export dynamics in the large basin. However, how land use change affects sediment export is still poorly understood. This study provided an integrated analysis of the relationships in a "three-level" chain linked as follows: "land use change → changes in sediment source and sediment delivery → sediment export change" for a better understanding. It used the InVEST sediment delivery ratio (SDR) model to analyze the Qiantang River Basin ($4.27*10^4$ km^2), China. Sediment export change was examined from the two perspectives: the effects of land use change on sediment source and on sediment delivery. Correlations between changes in individual land use types and changes in sediment source and sediment delivery were identified. The results indicated that sediment export reduced from 1.69 t ha^{-1} yr^{-1} in 1990 to 1.22 t ha^{-1} yr^{-1} in 2015 because of the decreased sediment source and a weakened sediment delivery function. In the study area, the conversions of cropland to urban land (urbanization) and bare land to forestland (afforestation) were found to make the major contributions to reductions in soil loss and SDR, respectively. Furthermore, soil loss change resulted in the decreases in total value of sediment export and SDR change caused a large-scale spatial change in sediment export. Our hotspot analysis revealed that the Wuxi River watershed should be targeted for priority conservation to optimize land use/cover for reducing sediment export. This study demonstrates the benefits of taking a comprehensive approach to analyze the processes associated with sediment export change. These allow to improve sediment management and promote aquatic ecosystem health by providing specific future land use recommendations, aimed at source treatment and delivery interception.

Keywords: Land use change; Soil loss; Sediment delivery ratio; Sediment export; InVEST model

1 Introduction

China has experienced drastic land use change caused by rapid urbanization and land use policies since the market-directed economic system was implemented in 1992 (Liu et at., 2014; Liu et al., 2018). This has had significant influences on sediment export to rivers because of the change in surface roughness, soil infiltration rate and hydraulic connectivity within watersheds (Fiener et al., 2011; Tang et al., 2011). Redundant sediment export reduces soil fertility, water and nutrient retention capacity, and increases suspended solids in water. These threaten ecosystem health and increase the risk of reservoir sedimentation, reducing reservoir performance and increasing costs (Vanacker et al., 2003; Keeler et al., 2012).

Sediment export describes the amount of onsite sediment source actually reaching the catchment outlet. It is determined by soil erosion from the land surface by rainfall–runoff as well as sediment delivery processes based on land connectivity (Bakker et al. 2008; Vigiak et al., 2012). It is also a function of land use as sediment transport capacities vary for different land use types (Van Rompaey et al., 2001). Therefore, the impact of land use change on sediment export can be divided into two parts: impacts on sediment source and on sediment delivery. Recent watershed studies indicated that sediment export is affected by land use change, soil and water conservation measures, and other anthropogenic activities (Walling and Fang, 2003; Kondolf et al., 2014; Wang et al., 2016). Most of these studies have emphasized the impacts of land use on soil erosion (sediment source) and sediment yield and have sought to quantify the impacts reducing soil erosion as a control on the sediment yield (Bakker et al., 2008; Fang, 2017; Romano et al., 2018). However, the impacts and contribution of land use changes on sediment delivery and export are still poorly understood. Few studies have considered the integrated "three-level" chain of "land use change → changes in sediment source and sediment delivery → sediment export change", that is the relationships between changes in land use, sediment export and sediment source and delivery (Alatorre et al., 2012; Shi et al., 2012). Research in these areas is necessary to fill gaps in knowledge and understanding of sediment export processes and thereby to better support sediment control.

Sediment source and sediment delivery (part 2 of the chain above) can be described as soil loss and sediment delivery ratio (SDR) for the quantitative analysis. The revised universal soil loss equation (RUSLE) by Renard et al. (1997) predicts soil loss owing to water erosion as the main sediment source (Yang et al., 2003; Lu et al., 2004; Sun et al., 2014). A number of studies have used RUSLE to examine soil loss driven by land use change varying C and P factors with the land use changes but keeping other factors constant (e.g. soil properties, topography and climatic features) (Erskine et al., 2002; Wei et al.; 2007). Some researchers found that soil loss was more sensitive to changes in some key land use types (such as forestland and cropland) than in other land use types (Feng et al., 2010; Rao et al., 2016), suggesting the need to investigate the varying contributions to soil loss of different land use changes. The SDR is the fraction of gross soil erosion that is transported to rivers from a given catchment in a given time interval (Lu et al., 2006). Although many studies have investigated on SDR considering its definition, contributing factors, calculation methods and measurements (Vigiak et al., 2012; Woznicki and Nejadhashemi, 2013; Wu et al., 2013), little work has examined the spatial variation in SDR and how it relates to land use change, and how changes in individual land use types contribute to change in SDR at the watershed scale.

In order to address these gaps, the InVEST (Integrated Valuation of Environmental Services and Tradeoffs) sediment delivery ratio (SDR) model was applied in this study. This model has been widely utilized in reservoir management and instream water quality maintenance to map the overland sediment generation and delivery to streams (Hamel et al., 2015; Hamel et al., 2017). This model has been shown to perform well after calibration with the observation data, and has been successfully used to estimate sediment retention services and describe the spatial distribution of sediment export (Sánchez-Canales et al., 2015; Jiang et al., 2016). The resulting outputs include maps of soil loss, SDR and sediment export which were used to analyze the relationships between land use and individual results. Such analysis support understanding of how different land use changes contribute to sediment source and sediment delivery, and therefore sediment export.

The Qiantang River Basin, located in southeast China, provides the strong soil conservation services because of current land use/cover and erosion controls. However, it is subject to high soil erosion risks due to the frequency of heavy rainfall and its mountainous terrain which require greater consideration in future land use planning (Rao et al, 2014). Since the 1990s, land use in the study basin has undergone dramatic changes because of urbanization and specific land use policies. In our study, we attempt to explain sediment export change caused by land use changes through analysis of changes in soil loss and SDR using the InVEST SDR model. The objectives of this study are as follows: (1) to trace the dynamics of soil loss, SDR and sediment export under different land use patterns from 1990 to 2015; (2) to analyze the relationships between soil loss, SDR and land use composition at the sub-watershed level (n = 763); (3) to evaluate the impacts of changes in individual land use types on soil loss and SDR and to identify the dominant contributors to changes in rates of soil loss and SDR; (4) to explain the effects of changes in soil loss and SDR on sediment export; (5) to propose practical recommendations for land use planning in support of improved watershed management.

2 Methods

2.1 Study area

The Qiantang River Basin was in Zhejiang Province. Zhejiang Province is located in the south of the Yangtze River Delta on the southeast coast of China and has a typical landscape composition of ~70% mountains, ~10% water and ~20% fields (Fig. 1). It is one of the most developed provinces in China and the Qiantang River is the largest river in the Province. The basin has an area of $4.27*10^4$ km^2 and is dominated by a typical subtropical humid monsoon climate. The annual average air temperature is about 17°C and the annual precipitation is about 1500 mm. In recent years, land use in the river basin has dramatically changed especially urban land, cropland and forestland because of intensive human activities and land protection policies (Fig. 2(a)). To explore the relationship between land use and sediment export, the basin was divided into 14 watersheds and 763 sub-watersheds using Hydrology tools in ArcGIS software based on a digital elevation model.

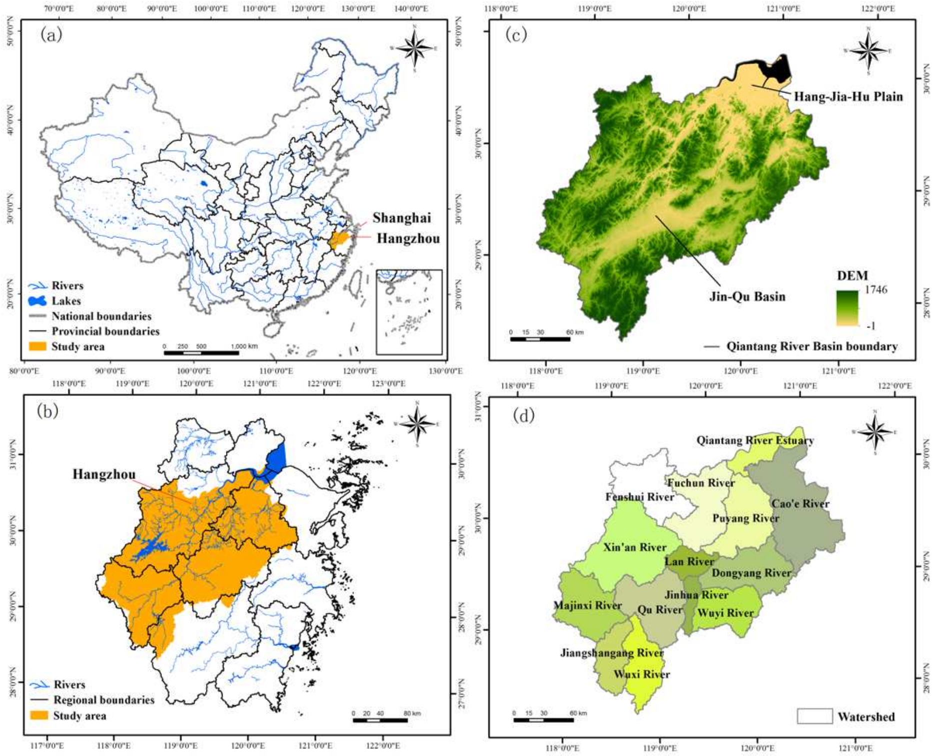

Fig. 1. Location of the study basin and watersheds: the location of the study basin in (a) China and (b) Zhejiang Province; (c) the elevation map of the study basin; (d) the distribution of the 14 watersheds.

2.2 Model description

Considering input data, complexity and model uncertainty, the InVEST SDR model was chosen. It calculates the sediment export by integrating a soil loss algorithm (Renard et al., 1997) with the sediment connectivity algorithm (Borselli et al., 2008), and generates maps of soil loss, SDR and sediment export as outputs. The main data requirements and sources in the model were represented in Table S1.

The sediment export ($t\ ha^{-1}\ yr^{-1}$) from a pixel i is given by:

$$Sed\_export_i = SL_i * SDR_i \qquad (1)$$

Where SL_i is the average amount of annual soil loss ($t\ ha^{-1}\ yr^{-1}$) on a pixel i; SDR_i is sediment delivery ratio for a pixel i. Ultimately, the total sediment yield in the catchment is the sum of sediment export from all pixels, which can be used to calibrate and validate the model.

Soil loss (SL) is computed with the RUSLE model (Renard et al., 1997):

$$SL = R * K * LS * C * P \qquad (2)$$

Where R is the rainfall erosivity $(MJ\ mm\ ha^{-1}\ h^{-1}\ yr^{-1})$; K is the soil erodibility $(t\ ha\ h\ ha^{-1}\ MJ^{-1}\ mm^{-1})$; LS is the slope length–gradient factor; C is the vegetation cover-management factor and P is the support practice factor.

SDR was calculated using the approach described in Vigiak et al. (2012), as a function of the hydrologic connectivity of the area derived from DEM. An index of connectivity, IC, describes the degree of hydrological connectivity of a pixel to stream. Here it was measured by its upslope contribution and flow path to the stream (Borselli et al., 2008). The SDR can be computed as:

$$SDR = \frac{SDR_{\max}}{1 + \exp(\frac{IC_0 - IC}{k_b})} \tag{3}$$

$$IC = \log_{10}\left(\frac{D_{up}}{D_{dn}}\right) \tag{4}$$

$$D_{up} = \bar{C}\bar{S}\sqrt{A} \tag{5}$$

$$D_{dn} = \sum_i \frac{d_i}{C_i S_i} \tag{6}$$

Where SDR_{max} is the maximum theoretical SDR, adopting a default value of 0.8 (Vigiak et al., 2012); IC_0 and k_b are calibration parameters; D_{up} is the upslope component; D_{dn} is the downslope component; $\bar{C}$ is the average C factor of the upslope contributing area; $\bar{S}$ is the average slope gradient of the upslope contributing area ($m\ m^{-1}$); A is the upslope contributing area (m^2); d_i is the average length of the flow path along the ith cell according to the steepest downslope direction (m); C_i and S_i are the C factor and the slope gradient of the ith pixel, respectively. The upslope contributing area and the downslope flow path is delineated from the D-infinity flow routing algorithm (Tarboton, 1997).

2.3 Parameters

2.3.1 Parameters in the RUSLE model

The calculation methods of parameters in RUSLE model were received from the related researches (Sheng at al., 2010; Rao et al., 2014; Kong et al., 2018) and were detailed in Text S1. First, rainfall erosivity (R), as the primary factor in the RUSLE model, describes the potential of rainstorms to cause soil erosion (Wischmeier and Smith, 1978; Zhang et al., 2002). The annual rainfall erosivity, a raster generated by the Kriging interpolation of data from 23 weather stations from 1990 to 2015, was calculated from the daily rainfall data using half-month rainfall erosivity model proposed by Zhang et al. (2002) (Text S1). This model was estimated using daily rainfall data and has been widely used in China (Xin et al., 2011; Sun et al., 2014; Yang and Lu, 2015).

Next, the soil erodibility factor (K) reflects the sensitivity of soils to water erosion due to soil properties (Zhang et al., 2008; Rao et al., 2014). In this paper, the soil erodibility factor value was derived from the revised erosion/productivity impact calculator (EPIC) model (Williams et al., 1984; Zhang et al., 2008) (Text S1).

The topographic factor (LS) captures the effect of slope length and slope gradient on soil erosion (Wischmeier and Smith, 1978) and can be computed from a DEM (McCool et al, 1997; Renard et al., 1997) (Text S1).

The vegetation cover factor (C) is sensitive to natural and anthropogenic activities and is critical to soil and water conservation (Wang et al., 2001). The value of C directly affects the value of soil loss and SDR. Based on the previous studies, the model by Cai et al. (2000) was applied for forestland, shrubland and grassland, and the method by Liu et al. (1999) was used for dry land (Text S1). While for water, paddy field, garden plot, urban land and bare land, values of 0, 0.1, 0.18, 0.01 and 0.7 were assigned, respectively (Cai et al., 2000; Yang et al., 2003; Rao et al. 2014).

The P factor describes the impact of support practices. It is the ratio of soil loss with contouring and strip cropping to that corresponding to losses under up-and-down-slope farming (Wischmeier and Smith, 1978). The P factor values for agricultural land vary widely in different regions because of different farming practices and geographical environments. Based on the study of the Southern Hillside Area of China (Chen et al. 2014), we assigned dry land, paddy field and garden plot with the values of 0.15, 0.4 and 0.18, respectively. For other land use types, P factor values from these studies (Yang et al., 2003; Teng, 2017) were applied.

2.3.2 Parameters in the SDR model

Threshold flow accumulation is used to extract streams from a DEM. The number of upstream cells that must flow into a cell before it is considered part of a stream (Sharp et al., 2018), was set to 1000 in this study similar to previous research (Zhong et al., 2013). The k_b and IC_0 determine the shape of the relationship between SDR and IC (Sharp et al., 2015). Vigiak et al. (2012) suggested that IC_0 is landscape independent and that the model is sensitive to k_b. Therefore, we set IC_0 as the default value of 0.5 and adjusted k_b according to the features of the study basin. The k_b was set as the value of 3.8 based on the previous studies (Hamel et al., 2015; Hamel et al., 2017) and the model calibration tests.

2.4 Validation

In this study, we validated the model results using the observation data and the previous researchers (Table S2). The observation data for the Qiantang River Basin covers two provinces, Anhui Province and Zhejiang Province, and thus we validated the results using annual average values. First, the downward trend of soil loss in the published data since the 1960s was consistent with the trend in this study. Next, the average value of resulting SDR was 0.1021 from 1990 to 2015, close to the published value of 0.11 from 1996 to 2005 (MWR, 2010a). The long-time average annual sediment yield in four gauging stations of study basin, Quzhou, Lanxi, Shangyu and Zhuji, were recorded as 1.91 t ha^{-1} yr^{-1}, 1.24 t ha^{-1} yr^{-1}, 1.05 t ha^{-1} yr^{-1} and 0.98 t ha^{-1} yr^{-1}, respectively, with an average value for the Qiantang River Basin of 1.18 t ha^{-1} yr^{-1} (measured over 2.44 km^2). The range of the resulting sediment export values, from 1.69 t/ha in 1990 to 1.22 t ha^{-1} yr^{-1} in 2015, agreed well with the actual value ranges. In addition, a good consistency of spatial distribution between the published soil erosion regions of Zhejiang Province and the identified high-risk regions in the results was found (Fig. S1). Thus, it was possible to confirm that the model and parameters used here were able to reliably simulate general sediment export and the model results were reasonable in the study basin.

3 Results

3.1 Land use change from 1990 to 2015

Fig. 2(a) shows that land use has undergone dramatic changes, with the main conversion of cropland to forestland and urban land from 1990 to 2015. The area of forestland accounted for about 60% of the total area (Fig. 3) and increased from 25431.83 km^2 to 26608.41 km^2, and the area of urban land rapidly expanded from 4.01% to 10.08% and increased by 2587.89 km^2 during the study period (Table S3). Conversely, cropland (paddy field and dry land) experienced a significant decline of 5.51%, including 1788.35 km^2 and 559.83 km^2 losses of paddy field and dry land, respectively. Some 17.86% and 74.54% of cropland area loss were converted to forestland and urban land (Table S4). Meanwhile, the areas of shrubland, grassland, water, garden plot, and bare land moderately decreased. According to the change area proportions in different periods, land use presented more drastic changes between 2000 and 2010 compared to the periods of 1990-2000 and 2010-2015 (Table S3). Spatially, land use changes mainly occurred in the plain area, including the Hang-Jia-Hu Plain and the Jin-Qu Basin (Fig. 2(a)).

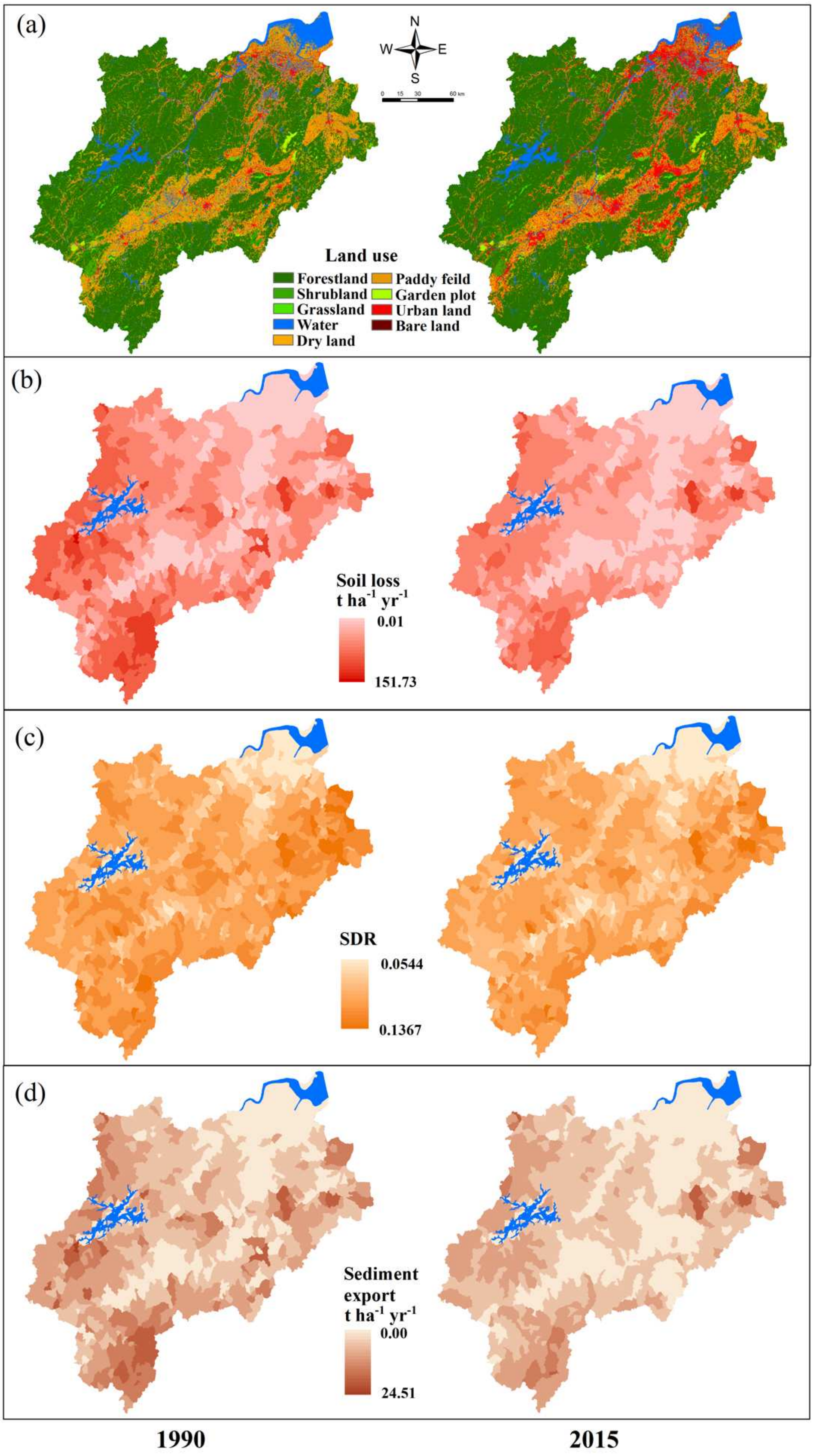
(a)
N
W
E
S
0 15 30 60 km
Land use
Forestland
Paddy feild
Shrubland
Garden plot
Grassland
Urban land
Water
Bare land
Dry land
(b)
Soil loss
t ha^{-1} yr^{-1}
0.01
151.73
(c)
SDR
0.0544
0.1367
(d)
Sediment
export
t ha^{-1} yr^{-1}
0.00
24.51
1990
2015

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Fig. 2. Spatial distribution of (a) land use; the average values of (b) soil loss, (c) SDR and (d) sediment export, for each sub-watershed between 1990 and 2015.

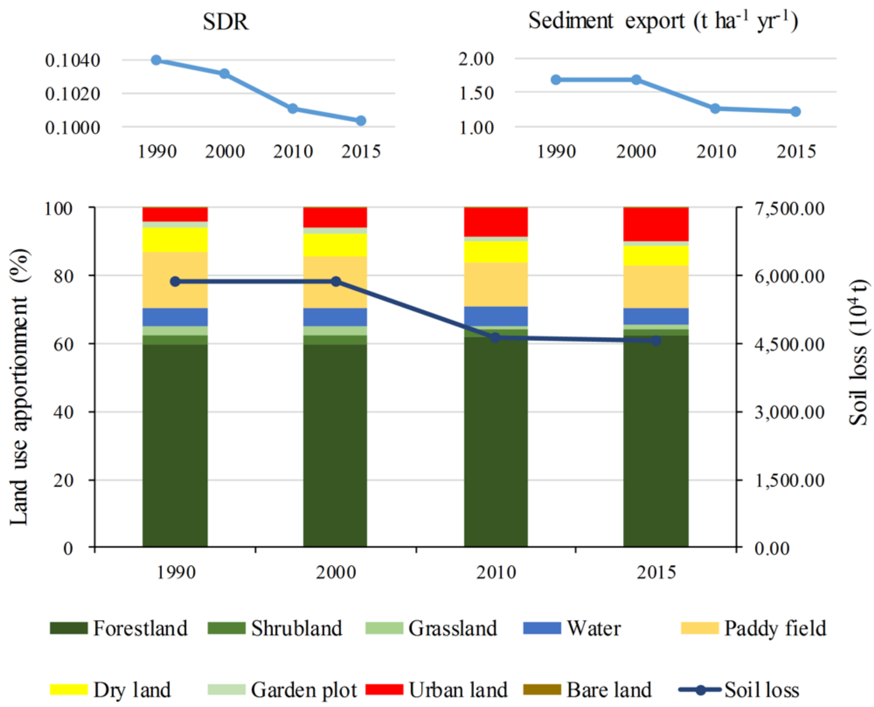

Fig. 3. Land use composition and the changes in soil loss, SDR and sediment export from 1990 to 2015 in the study basin.

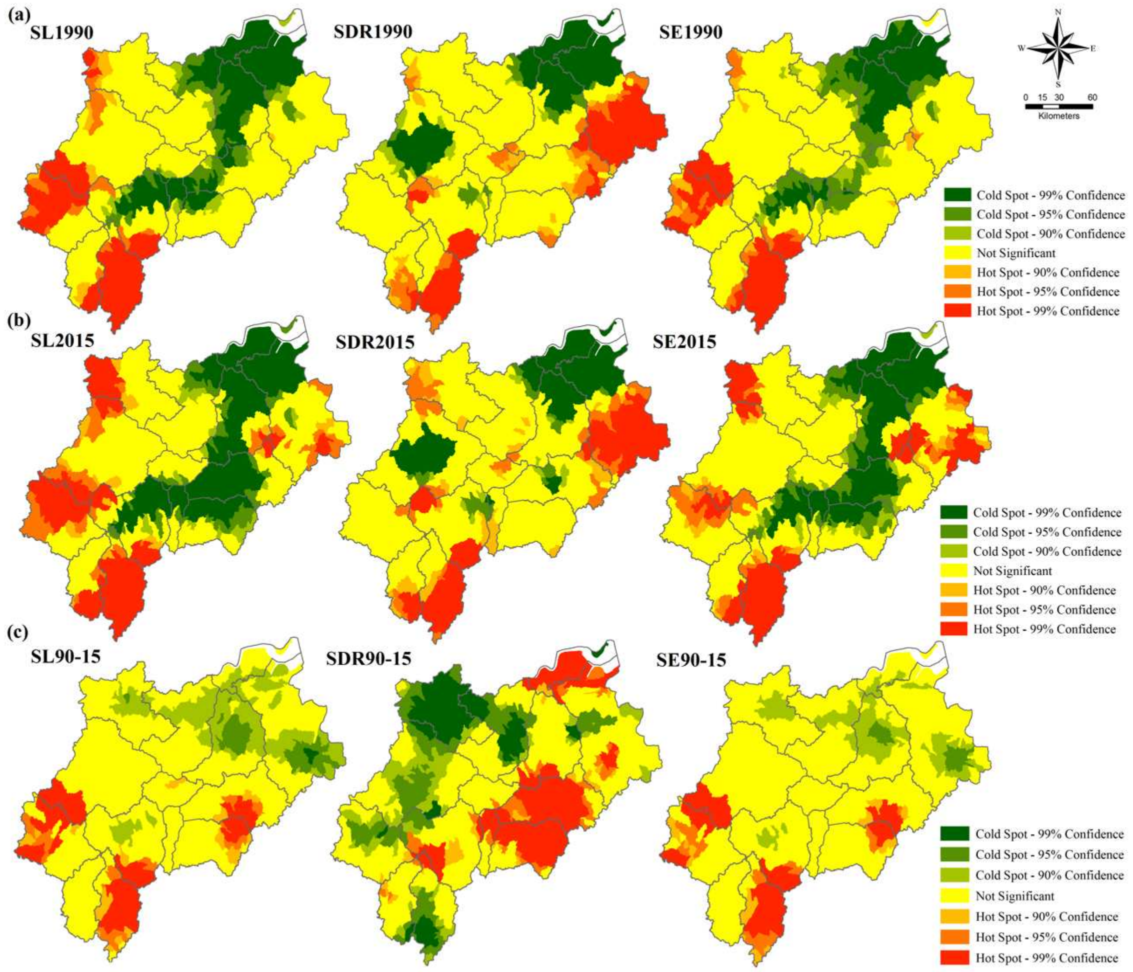

Fig. 4. Spatial distribution of hotspots and cold spots of soil loss (SL), SDR, and sediment export (SE) in (a) 1990 and (b) 2015, and (c) differences between the average values of 736 sub-watersheds in 1990 and these in 2015.

3.2 Analysis of "land use change → changes in soil loss and SDR"

3.2.1 Changes in soil loss and SDR from 1990 to 2015

The average soil loss decreased from 13.79 t ha^{-1} yr^{-1} in 1990 to 10.70 t ha^{-1} yr^{-1} in 2015. The total values of soil loss were $5880.24*10^4$ t, $5874.41*10^4$ t, $4642.09*10^4$ t and $4563.66*10^4$ t in 1990, 2000, 2010 and 2015, respectively (Table 1). The average soil loss from bare land was highest among all land use types and had the largest drop from 686.10 t ha^{-1} yr^{-1} in 1990 to 328.69 t ha^{-1} yr^{-1} in 2015. Meanwhile, because the area of bare land reduced from 120.38 km^2 to 14.58 km^2, the total soil loss from bare land decreased by $778.01*10^4$ t. This decrease accounted for approximately 60% of the total decreased soil loss (Table 1). Some 68.76% of bare land was converted to forestland (Table S4) and this conversion resulted in a decrease of $700.83*10^4$ t in soil loss. In addition, the soil loss from cropland and garden plot reduced by $255.21*10^4$ t and $301.27*10^4$ t (Table 1). Conversely, the total soil loss from forestland and urban land increased slightly by $99.24*10^4$ t and

29.82*10^4 t, and the average value of soil loss from urban land increased from 0.98 t ha^{-1} yr^{-1} to 1.08 t ha^{-1} yr^{-1} (Table 1).

Using the Hot Spot Analysis (Getis-Ord Gi∗) embedded in ArcGIS software, we found that hotspots of soil loss were mostly clustered in the hilly sub-watersheds in the Wuxi River watershed, the Majinxi River watershed and the Xin'an River watershed in 1990, and in the additional sub-watersheds in the Fenshui River watershed in 2015 (Fig. 4). In contrast, most cold spots were identified in flat terrains, such as the Hang-Jia-Hu Plain and Jin-Qu Basin, and spread to the peripheral sub-watersheds due to the conversion of cropland to urban land in the plain from 1990 to 2015. Soil loss had a pronounced decrease (hotspots) in the Wuxi River watershed, the junction of the Dongyang River watershed and Wuyi River watershed and the junction of the Majinxi River watershed and Xin'an River watershed from 1990 to 2015 (Fig. 4(c)). Relatively, the cluster of the cold spots of soil loss change (where soil loss increased) was insignificant and distributed in the eastern sub-watersheds (Fig. 4(c)).

The SDR values in 1990, 2000, 2010 and 2015 were 0.1040, 0.1031, 0.1011 and 0.1003 respectively across the whole basin (Fig. 3). The SDR of all land use types declined except for that of grassland (Fig. S2). The SDR for bare land was the highest and most stable, from 0.1497 in 1990 to 0.1495 in 2015. In contrast, the SDR for urban land was the lowest and obviously reduced from 0.0820 in 1990 to 0.0798 in 2015 (Fig. S2). Spatially, the areas with the highest SDR were near rivers in the hills, and the areas with the lowest SDR were located in the large plains.

Fig. 4 indicates that the sub-watersheds with low SDR clustered in the Xin'an River watershed and Hang-Jia-Hu Plain in 1990 and 2015, and those with high SDR clustered in the upstream regions of the Cao'e River watershed and Wuxi River watershed (Fig. 4(a)(b)). Moreover, Fig. 4(c) shows that the sub-watersheds with a large decrease (hotspots) in SDR were found in the Qiantang River estuary, Dongyang River, Wuyi River and Jinhua River. In these areas, a considerable amount of cropland was occupied by urban land. Simultaneously, the cold spots were identified in the west of the basin, including the Fenshui River watershed, Xin'an River watershed and Wuxi River watershed.

Table 1

Average and total values of soil loss and sediment export for different land use types from 1990 to 2015 in the study area.

	Year	Forestland	Shrubland	Grassland	Paddy field	Dry land	Garden plot	Urban land	Bare land	A/T
ASL	1990	11.36	14.27	11.93	3.39	32.60	96.91	0.98	686.10	13.79
	2000	11.33	14.25	12.37	3.54	33.53	82.97	0.96	747.20	13.77
	2010	11.26	14.42	15.59	3.60	30.81	60.35	1.06	386.07	10.88

	2015	11.23	14.44	16.91	3.63	31.64	61.19	1.08	328.69	10.70
TSL	1990	2888.85	161.74	144.02	236.93	1008.47	597.58	16.72	825.94	5880.24
	2000	2894.00	161.85	131.51	224.71	960.73	639.57	23.53	838.51	5874.41
	2010	2972.44	132.29	84.24	201.84	827.95	310.33	38.83	74.20	4642.12
	2015	2988.09	118.33	76.28	188.62	801.57	296.31	46.54	47.93	4563.66
ASE	1990	1.17	1.55	1.38	0.44	4.73	13.25	0.09	105.73	1.69
	2000	1.17	1.54	1.42	0.46	4.86	11.36	0.09	115.05	1.69
	2010	1.15	1.54	1.80	0.47	4.50	8.75	0.10	58.25	1.26
	2015	1.15	1.54	1.94	0.47	4.60	8.77	0.10	50.18	1.22
TSE	1990	298.15	17.54	16.67	31.05	146.36	81.68	1.61	127.28	720.34
	2000	298.46	17.53	15.15	29.36	139.25	87.58	2.25	129.10	718.67
	2010	303.94	14.16	9.73	26.30	121.07	44.98	3.69	11.19	535.07
	2015	305.16	12.62	8.77	24.43	116.45	42.46	4.44	7.32	521.65

Note: ASL: average soil loss (t ha^{-1} yr^{-1}); TSL: total soil loss (t); ASE: average sediment export (t ha^{-1} yr^{-1}); TSE: total sediment export (t); A/T: average value or total value.

3.2.2 Relationships between land use composition, soil loss and SDR

We analyzed the relationships in the 763 sub-watersheds between land use composition and soil loss and between land use composition and SDR using Pearson's correlation analysis (Table 2). The proportion of forestland was strongly and positively correlated with soil loss and SDR. Conversely, the proportion of urban land was strongly and negatively correlated with soil loss and SDR. There was a significantly negative correlation between the proportion of water and SDR. The proportion of cropland had a strong and negative correlation with soil loss (paddy field was stronger than dry land), in contrast to a weakly negative correlation with SDR. The results revealed that soil loss was relatively low in sub-watersheds dominated by urban land and cropland, and high in sub-watersheds with high proportions of bare land and forestland. SDR was relatively low in sub-watersheds dominated by urban land and water but was high in sub-watersheds with high areas of forestland.

Table 2

Pearson correlation coefficients for the relationship between land use proportion and the average values of soil loss (SL) and SDR in 763 sub-watersheds from 1990 to 2015.

		Forestland	Shrubland	Grassland	Water	Paddy field	Dry land	Garden plot	Urban land	Bare land	Cropland
1990	SL	0.47**	0.14**	-0.13**	-0.32**	-0.47**	-0.22**	0.21**	-0.43**	0.55**	-0.43**
	SDR	0.41**	0.07	-0.01	-0.83**	-0.16**	0.05	0.20**	-0.46**	0.13**	-0.09*
2000	SL	0.47**	0.15**	-0.12**	-0.32**	-0.44**	-0.21**	0.17**	-0.43**	0.57**	-0.40**

	SDR	0.47**	0.07*	-0.03	-0.81**	-0.13**	0.03	0.15**	-0.55**	0.12**	-0.08*
2010	SL	0.58**	0.08*	0.02	-0.37**	-0.44**	-0.22**	0.22**	-0.58**	0.12**	-0.41**
	SDR	0.50**	0.04	0.07	-0.78**	-0.11**	0.08*	0.11**	-0.58**	-0.01	-0.03
2015	SL	0.60**	0.07	0.04	-0.36**	-0.43**	-0.21**	0.21**	-0.61**	0.12**	-0.40*
	SDR	0.54**	0.05	0.07	-0.76**	-0.12**	0.05	0.10**	-0.60**	-0.01	-0.06

* P<0.05 (two-tailed)

** P<0.01(two-tailed)

3.2.3 Relationships between changes in individual land use types and changes in soil loss and SDR

Land use changes resulted in soil loss and SDR changes from 1990 to 2015. To further quantify the contribution of changes in individual land use types, we examined the changes in soil loss and SDR against changes in land use proportion (Fig. 5). Strong positive correlations were found between changes in bare land and soil loss (R^2=0.68, P<0.001) and negative correlations were found between changes in forestland and soil loss (R^2=0.26, P<0.001). Conversely, there was no significant relationship between SDR change and forestland change (R=0.052, P=0.151) (Fig. S3). SDR changes showed more significant correlations with changes in land use more closely associated with human activities, such as cropland (R^2=0.76, P<0.001) and urban land (R^2=0.61, P<0.001). A weak relationship was found between soil loss and cropland changes (R=-0.095, P=0.009) (Fig. S4). As a result, soil loss decreased sharply in sub-watersheds where bare land was converted to forest. Similarly, SDR also showed a dramatic decline in sub-watersheds where cropland was converted to urban land.

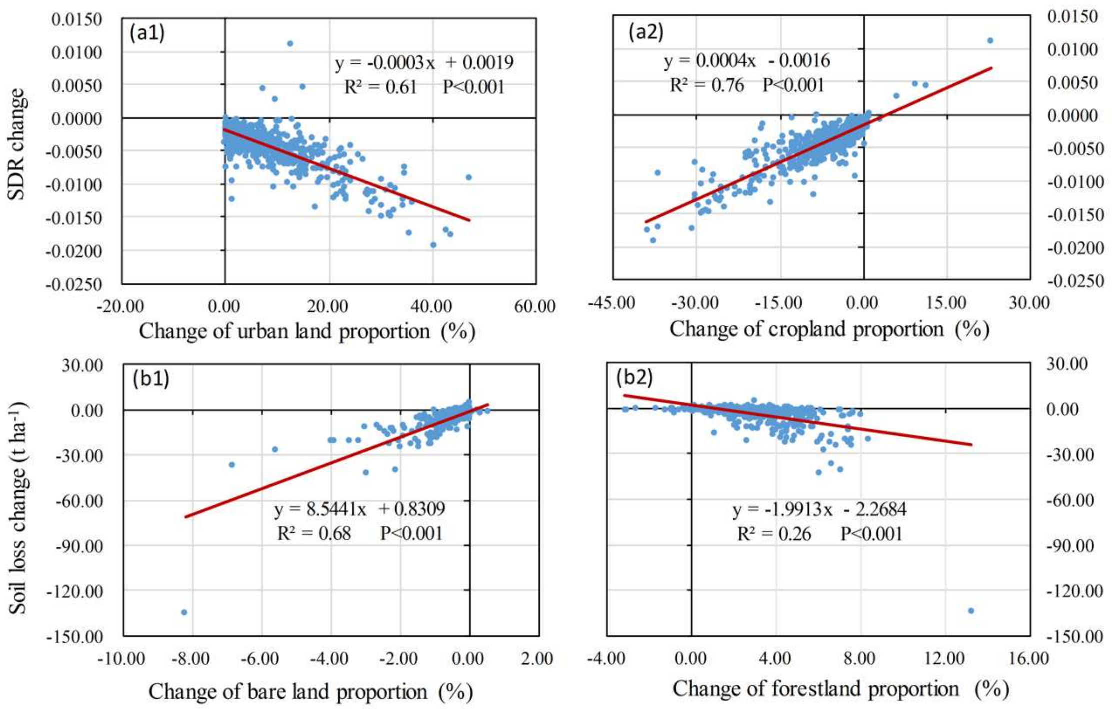

Fig. 5.

Relationships between changes in the area proportion of different land use types and changes in the average values of soil loss and SDR in the sub-watersheds: (a1) SDR vs urban land; (a2) SDR vs cropland; (b1) soil loss vs bare land; (b2) soil loss vs forestland.

3.3 Analysis of "changes in soil loss and SDR → sediment export change"

3.3.1 Sediment export change from 1990 to 2015

Sediment export had a significant decline from 1990 to 2015 (Fig. 2). The average sediment export decreased from 1.69 t ha^{-1} yr^{-1} to 1.22 t ha^{-1} yr^{-1}, and the total sediment export reduced from 720.34*10^4 t to 521.65*10^4 t, with approximately 60% of this decline from the loss of bare land (Table 1). The top three reductions in sediment export caused by land use transitions were the conversions of bare land to forestland, garden plot to forestland and dry land to forestland. These accounted for 54.58%, 19.18% and 14.72% of the total decrease, respectively (Table S5). Conversely, the top three increases in sediment export were 17.27*10^4 t, 11.17*10^4 t and 4.00*10^4 t, as a result of conversions of paddy field to dry land, grassland to dry land and paddy field to garden plot, respectively (Table S5). The results implied that decreases in bare land greatly contributed to the reductions of sediment export. In contrast, increases in agricultural land, such as cropland and garden plot, were found to increase sediment export in the study basin.

Spatially, the cold spots of sediment export were clustered in the low-slope areas and the cold spots and hotspots in 2015 were more widely spread than those in 1990 (Fig. 4(a) (b)). The Wuxi River watershed was found to contain most hotspots of sediment export in 1990 and in 2015, as well as most hotspots of sediment export change from 1990 to 2015. That is, this watershed dramatically declined while having high sediment export. In addition, there were some hotspots of sediment export in the Fenshui River watershed, the junction of the Xin'an River watershed and Majinxi River watershed, and Cao'e River watershed (Fig. 4(a) (b)). Rather, the cluster of the cold spots of sediment export change was relatively insignificant (Fig. 4(c)).

3.3.2 Effects of changes in soil loss and SDR on sediment export change

From 1990 to 2015, increases and decreases of soil loss accounted for 2.43% and 9.83% respectively of the whole watershed area and the increases and decreases of SDR accounted for 10.26% and 55.21% (Fig. 2). The area of increased sediment export accounted for 9.40% of the whole watershed area, mainly due to 7.11% of the total area where SDR increased and soil loss was unchanged. The area of decreased sediment export accounted for 52.58% of the whole watershed area, due to 43.17% of the total area where SDR decreased and soil loss was unchanged (Fig. 6). That is, SDR change resulted in a large-scale change in sediment export in the study area.

Alternately, the reduction in the average value of soil loss (22.41%) was much greater than that of SDR (3.56%) and the average value of sediment export reduced by 27.81%. Fig. 4 shows that the distributions in hotspots (high values) and cold spots (low values) of sediment export and sediment export change were highly consistent with those of soil loss.

Generally, soil loss change resulted in a significant decrease in the total sediment export and related to the magnitude of the values of sediment export change in the sub-watersheds.

Overlay analysis showed that there were low values of soil loss, SDR and sediment export in the Hang-Jia-Hu Plain in 1990 and 2015, while high values were found in the Wuxi River watershed and the downstream regions of the Qu River watershed and Jiangshangang River watershed (Fig. 4(a) (b)). Significant decreases (hotspots) in soil loss, SDR and sediment export were found at the junction of the Dongyang River watershed and Wuyi River watershed (Fig. 4(c)).

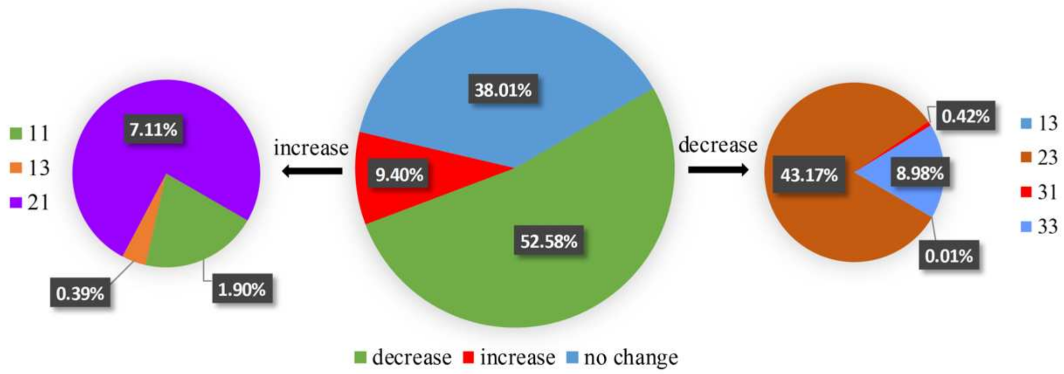

Fig. 6. The area proportions of sediment export changes and their compositions from 1990 to 2015. The values of 1, 2 and 3 represent increases, no change and decreases from 1990 to 2015, respectively. The first of the ten digits in the legends represents the change in soil loss, and the second represents the change in SDR.

4 Discussion

4.1 Evaluation on sediment export in the Qiantang River Basin

This study highlighted the “three-level” relationships throughout the process of sediment transport dynamics, and distinguished the different effects of sediment source and sediment delivery on sediment export reduction under the rapid land use change in the Qiantang River Basin. Urbanization and afforestation, which were in accordance with the land use policies, were found to be the dominant causes of reduction in SDR and soil loss, respectively, and greatly encouraged a decrease in sediment export. This finding was consistent with some similar studies, which also showed a decline in sediment yield due to land use change and soil conservation projects (Chu et al., 2009; Liu et al., 2013). Reduced sediment yield from major rivers in China was found and its main driving factor was converted from dam construction to conservation measures (especially the Grain for Green Program) after 1999 (Li et al., 2018). Since the 1950s, sediment loads from rivers in South China, including the Pearl River, Min River and Qiantang River, reduced by 44%~58% (Hu et al., 2010). Among the three rivers, the Qiantang River had the least soil loss and SDR, but its soil loss was far more than the soil loss tolerance in China (2~10 t ha^{-1} yr^{-1}) (MWR, 2010a). In addition, main anthropogenic drivers of the decreased sediment loads from these rivers were different since the 1990s. ~90% of reduction in the Pearl River Basin was caused by dam construction (Wu et al., 2012). A clear decrease in sediment transport in the Min River could be attributed to reservoir construction and sand mining (Liu et al., 2008; Hu et al., 2010). However, reduction in the Qiantang River Basin primarily benefited from soil conservation practices, especially the increased forestland (Zhang et al., 2015).

4.2 Drivers of land use change from 1990 to 2015

In this study, the main land use changes were the conversion of cropland to forestland and urban land, and presented different change rates in three periods from 1990 to 2000, 2000 to 2010 and 2010 to 2015, which can be described as a process of "start-acceleration-slow" (Table S3). Land use dynamics have been primarily driven by economic development and land use policies in China (Liu et al., 2014). In the first decade, because the socialist market economic system launched in 1992, urbanization was comprehensively promoted and the Afforestation and Greening Project was on highlight era stage, land use change was in the "start" stage. In the next decade, land use change accelerated with a rate of 8.01% (Table S3) and the proportions of forestland and urban land greatly increased due to accelerating urbanization and ecological restoration projects. In recent years, as the Farmland Protection System was constantly revised and comprehensively implemented, the transformation of agricultural land to non-agricultural land gradually reduced, the growth of forestland from cropland slowed but urbanization continued at the same rate. Under these land use change patterns, this study found a reduction in sediment source and delivery, resulting in a decline in sediment export. However, as the Permanent Prime Farmland Protection Policy and other Farmland Protection policies are implemented, resulting in different patterns of land use change, it may difficult to predict future sediment export.

4.3 Discussion on the "three-level" relationships

The links between land use change, soil loss and SDR are built on the C and P factors. Variation in the values and spatial distribution of C factor directly alters the soil loss, and the reductions of C factor result in decreases in IC related to SDR values. The P factor has an impact on soil loss but has no direct correlation with SDR, with smaller C and P factors related to stronger controlling sediment export (Table 3).

The distributions of different land use types follow a number of general trends: forestland tends to be found in the hilly and steep areas, cropland and urban land are typically found in in the plain areas. Sun et al. (2014) noted that the hilly and gully regions with higher LS values are topographically prone to erosion, which largely explains the relationship between soil loss and land use (Table 2) and distribution of the cold spots and hotspots of soil loss (Fig. 4). In addition, bare land with high value of C factor and that tends to be found in areas with terrible topographic and soil environments, was associated with much large soil losses in the sub-watersheds than losses from other land use types (Table 1). As a result, changes in bare land resulted in large fluctuations of soil loss in the sub-watersheds (Fig. 5(b1)). Increases in forestland were comparatively small because of the original dominant forestland proportion

(Fig. 1). As a result, reduction in soil loss from increased forestland was not as remarkable as that from decreased bare land in the sub-watersheds (Fig. 5(b2)). Because the SDR of water was the value of 0, sub-watersheds with larger proportion of water had lower SDR, such as Xin'an River watershed. According to the study by Borselli et al. (2008), SDR depends on the index of connectivity calculated from C factor, slope, and contribution area as well as flow path. Both urban land and cropland near rivers had short flow paths and low slope values, but lower SDR in urban land and higher SDR in cropland were found that were caused by the large differences in C factor (Fig. S2). Thus, the proportion of cropland had a less negative correlation with SDR than urban land in the sub-watersheds (Table 2). Large-scale agricultural planting may result in a much better connectivity caused by a degraded landscape, surface stoniness and raiding channels in the stream network (Beguería et al., 2005). Further, the study by Alatorre et al. (2012) also showed an 84% increase in SDR when the study area was occupied by annual crops compared with no crops, implying the close positive correlation between SDR and cropland (Fig. 5(a2)). It was proved that the upstream regions of Cao'e River with developed agriculture surrounded by mountains had a high SDR. Urban environments have a very low connectivity because of their low slopes and artificial surface providing a stronger barrier to sediment transport than other land use types (Borselli et al., 2008, Sharp et al., 2015). Therefore, watersheds with increased urban land from cropland, such as Qiantang River Estuary, made a great contribution to the decrease in SDR (Fig. 5(a1)).

SDR was found to be more stable in the large basin than soil loss when the effects of soil loss and SDR on sediment export are compared. SDR can be affected by the peripheral land use change in a given area contributing to the spatial connectivity of IC, therefore, the impact from it is related to the spatial scale of sediment export change. However, soil loss, as the main sediment source, is limited by the scope of land use change and fluctuates with land use/cover change. Therefore, its influence controls the total value of sediment export to rivers.

4.4 Suggestions

Significant progress and improvement have been made in sediment retention services with the development of soil and water conservation projects and land policies in Zhejiang (MWR, 2010b). The government invested 45.80 million Yuan to support the National Soil and Water Conservation Key Project in Zhejiang, including regulating slope drainage systems, afforestation on hills and converting sloping fields into terraces, by the Zhejiang Bulletin of Soil and Water Conservation (2015). Effective engineering and biological measures are important for sediment export control (Lin et al., 2008; Feng et al., 2010). Boix-Fayos et al. (2008) also suggested that land use changes can have important long-term effects on sediment yield with no side-effects. Therefore, it is also important to define comprehensive land use plans as well. Based on the results of this study, two land use types, forestland and bare land, can be used to control the sediment sources. It is important to improve vegetation coverage and quality, protecting the ecological forestland and greening bare land. Alternatively, the strong correlation between SDR change and changes in cropland and urban land suggests additional ways to control the sediment delivery. As the Farmland Protection Policy, means that reducing SDR by converting cropland to urban land and forestland is impractical, we draw lessons from recent land development policies and identify potentially beneficial measures. These included turning dry land to paddy fields and building ecological villages and towns in low-slope hilly regions. Additionally, there are opportunities for environment treatments to benefit areas with high SDR that are located near rivers in the hilly environments. According to the hotspot analysis, as the overlay areas of high values of soil loss, SDR and sediment export, the Wuxi River watershed should be listed as the key areas for future treatment. At the same time, these areas have large potential in improving sediment retention (Fig. 4(c)). However, the Hang-Jia-Hu Plain could be considered as a low-risk area. In the future, land use should be optimized in the upstream regions with severe sediment export, and more soil and water conservation projects implemented in those areas near rivers.

Table 3

Average values of C factor, P factor, CP and IC from 1990 to 2015 in the study basin.

	1990	2000	2010	2015
C	0.0559	0.0531	0.0473	0.0449
P	0.6218	0.6361	0.6548	0.6652
CP	0.0231	0.0225	0.0195	0.0188
IC	-6.7752	-6.8138	-6.8997	-6.9302

5 Conclusions

This study provided an effective method to evaluate sediment yield in a large basin. Based on the analysis of the “three-level” relationships, we explored the reduced sediment export (from 1.69 t ha^{-1} yr^{-1} in 1990 to 1.22 t ha^{-1} yr^{-1} in 2015) from the two perspectives of the decreased sediment source and a weakened sediment delivery function, using InVEST SDR model in the Qiantang River Basin. Urbanization and afforestation made the main contributions to decreases in soil loss and SDR, respectively. Furthermore, soil loss change and SDR change had strong effects on the magnitude of the value and the spatial scale of the sediment export change, respectively. In order to cope with new patterns of land use change, driven by continuous urban expansion, strict farmland protection policies and ecological protection projects in the future, this study identified a number of practical suggestions related to land use policies (e.g. building ecological villages and towns in low-slope hilly regions, turning dry land to paddy fields and implementing ecological forest protection) to improve the sediment management and aquatic ecosystems in the study basin.

However, some limitations were also highlighted here for guiding future studies. Predicting sediment yield remains challenging even for state-of-the-art models since sediment sources are diverse. In this study, the RUSLE model was used to measure the main sediment source from water erosion and thus the contribution from additional sediment sources might not be taken into account in the results. That is to say, this method is defective to be applied in some basins where are dominant by other sediment sources, such as mass erosion. What’s more, predicting sediment yield may be impacted by the DEM quality (resolution). Further analysis is needed to explore the sensitivity of sediment yield to DEM with a complicated topography in the Qiantang River Basin.

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Author Contributions

Mengmeng Zhou, Jinsong Deng, Yi Lin and Ke Wang conceived and designed this study; Marye Belete, Lingyan Huang and Muye Gan collected data; Mengmeng Zhou analyzed data and wrote this paper; Alexis Comber and Jinsong Deng revised the manuscript; all authors read and approved the final manuscript.

Appendix A. Supplementary data

Supplementary data associated with this article can be found in the online version. Text S1, Table S1−5 and Figure S1−4.

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Supplementary data

Text S1 Parameters in the RUSLE model

Rainfall erosivity factor (R):

R, as the primary factor in the RUSLE model, reflects the ability of rainwater to strip, move, and wash away soil under rainfall conditions, and describes the potential of rainstorms to cause soil erosion. The half-month rainfall erosivity model by Zhang et al. (2002) is described as follows:

$$R = \alpha \sum_{j=1}^{k} (P_j)^{\beta} \tag{1}$$

$$\beta = 0.8363 + \frac{18.144}{\overline{p}_{d12}} + \frac{24.455}{\overline{p}_{y12}} \tag{2}$$

$$\alpha = 21.586\beta^{-7.1891} \tag{3}$$

Where R is the half-month rainfall erosivity *(MJ mm ha^{-1} h^{-1} yr^{-1})* and P_j is the effective rainfall for day j in one half-month of k days. If the actual rainfall is greater than the threshold value of 12 *mm*, P_j is equal to the actual rainfall, otherwise, P_j is equal to zero (Xie et al., 2000). The terms α and β are the undetermined parameters; $\overline{p}_{d12}$ is the average daily rainfall that is greater than 12 *mm* and $\overline{p}_{y12}$ is the annual average rainfall for days with rainfall greater than 12 *mm*.

Soil erodibility factor (K):

The soil erodibility factor (K) measures the susceptibility of soil particles to detachment and transportation during rainfall and runoff. The revised erosion/productivity impact calculator (EPIC) model (Williams et al., 1984; Zhang et al., 2008):

$$K_{EPIC} = \left\{0.2 + 0.3\exp\left[0.0256SAN(1 - SIL/100)\right]\right\} * \left(\frac{SIL}{CLA + SIL}\right)^{0.3} * \left[1.0 - \frac{0.25C}{C + \exp(3.72 - 2.95C)}\right] * \left[1.0 - \frac{0.7SN_1}{SN_1 + \exp(-5.51 + 22.9SN_1)}\right] \tag{4}$$

$$K = (-0.01383 + 0.51575K_{EPIC}) * 0.1317 \tag{5}$$

Where K_{EPIC} and K is the soil erodibility factor $(t\ ha\ h\ ha^{-1}\ MJ^{-1}\ mm^{-1})$ before and after revision, respectively; *SAN*, *SIL*, *CLA* and *C* are the mass percentages of sand, silt, clay, and organic carbon, respectively; and SN_1 is equal to $1\text{-}SAN/100$. In equation (5), the value 0.1317 is the conversion factor from US units to SI units. Because the particle size classification standard used in the Second Soil Census data in China is based on the international system of soil texture, it is necessary to use logarithmic linear interpolation to transform the soil particle size data from the international system to the US system (Lu and Shen, 1992; Cai et al., 2003).

Topographic factor (LS):

The LS calculation was based on DEM (McCool et al, 1997; Renard et al., 1997) :

$$L = \left(\frac{\lambda}{22.13}\right)^m \tag{6}$$

$$S = \begin{cases} 10.8\sin\theta + 0.03 & \theta < 9\% \\ 16.8\sin\theta - 0.50 & \theta \geq 9\% \end{cases} \tag{7}$$

Where L is the slope length factor; S is the slope steepness factor; m is the slope length index and is acquired from the table by McCool et al. (1997); θ is the slope gradient (°); and λ is the slope length (m). The parameters are computed based on the digital elevation model.

Vegetation cover factor (C)

For forestland, shrubland and grassland (Cai et al., 2000):

$$C = \begin{cases} 1 & v = 0 \\ 0.6508 - 0.3436\lg v & 0 < v \leq 78.3\% \\ 0 & v > 78.3\% \end{cases} \tag{8}$$

For dry land (Liu et al., 1999):

$$C = 0.221 - 0.595\log v \tag{9}$$

Where v is the vegetation cover factor of land use types. It is noteworthy that v is calculated as a proportion in equation (8) and as a percentage in equation (9).

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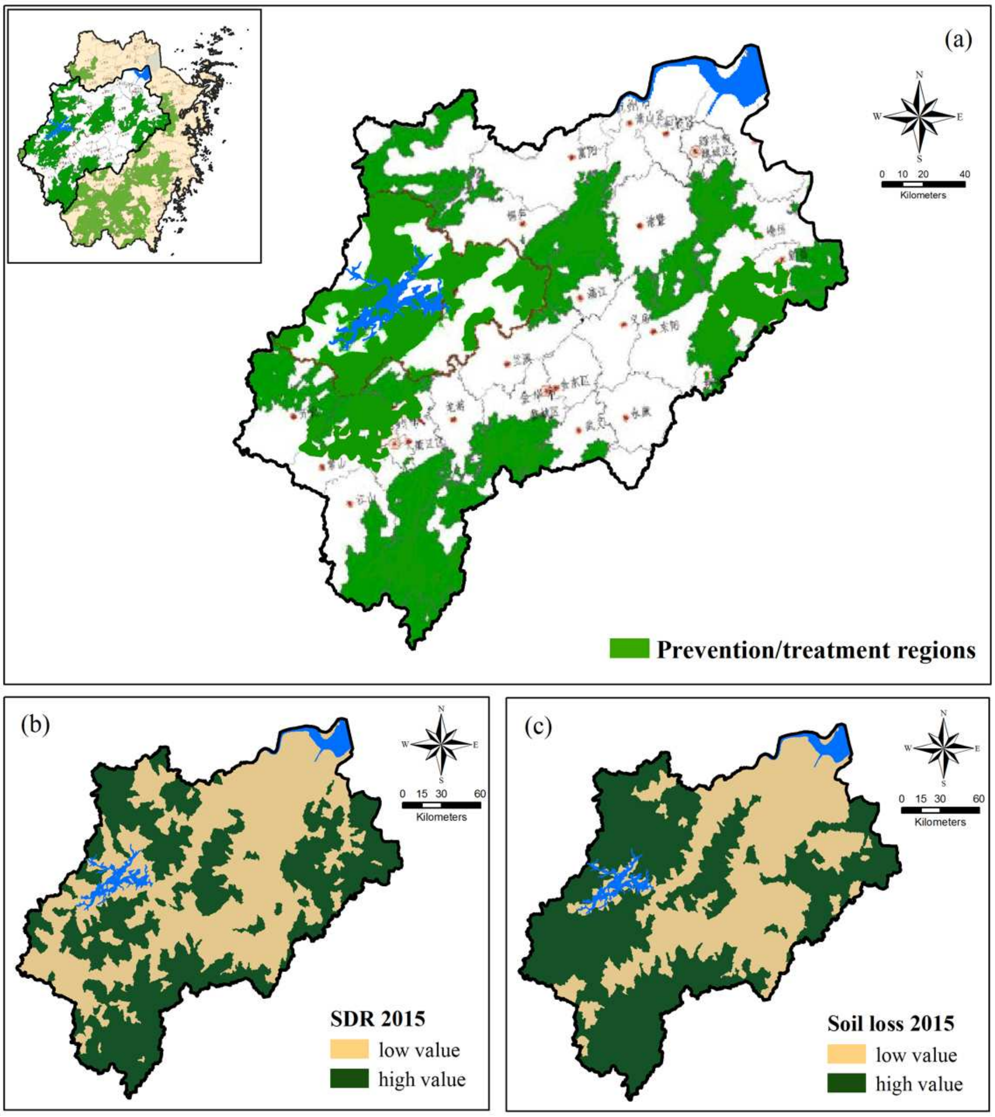

Fig. S1. (a) Published data: key prevention/treatment regions for soil erosion in Zhejiang Province and Qiantang River Basin; model results: spatial distribution of (b) SDR and (c) soil loss in 2015.

Note: Fig S1(a) was extracted from a published figure. (Zhejiang Water Resources Department, Zhejiang Development and Reform Commission, 2015, Announcement on key prevention/treatment regions and basic conditions of soil erosion in Zhejiang Province (in chinese).)

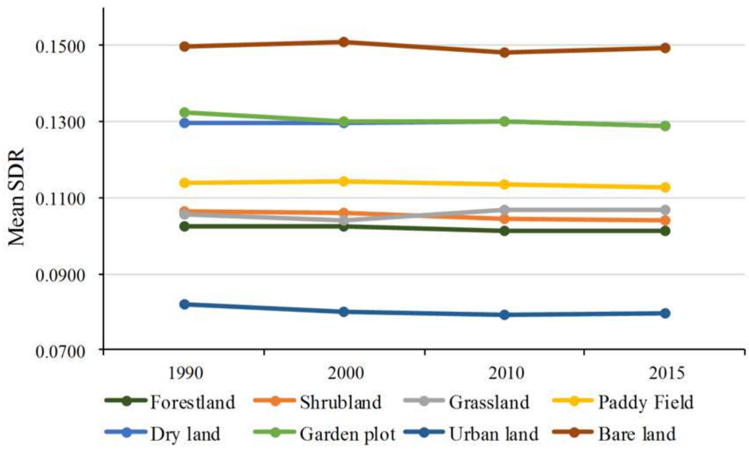

Fig. S2. The SDR of different land use types from 1990 to 2015 in the study basin.

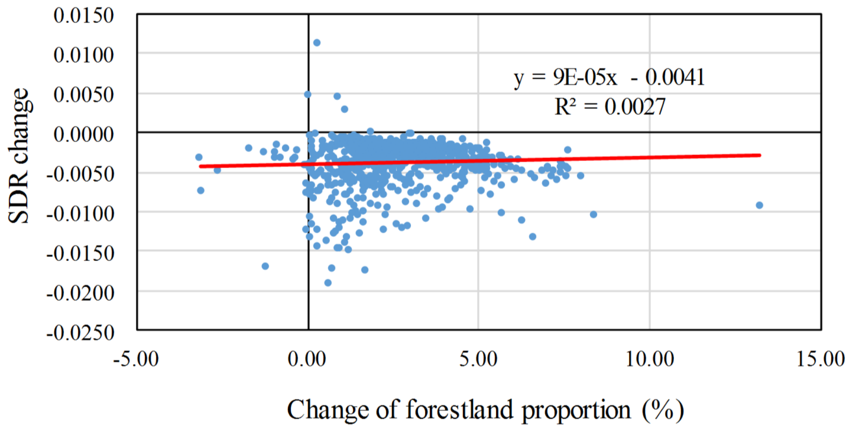

Fig. S3. Relationship between change of forestland proportion and SDR change.

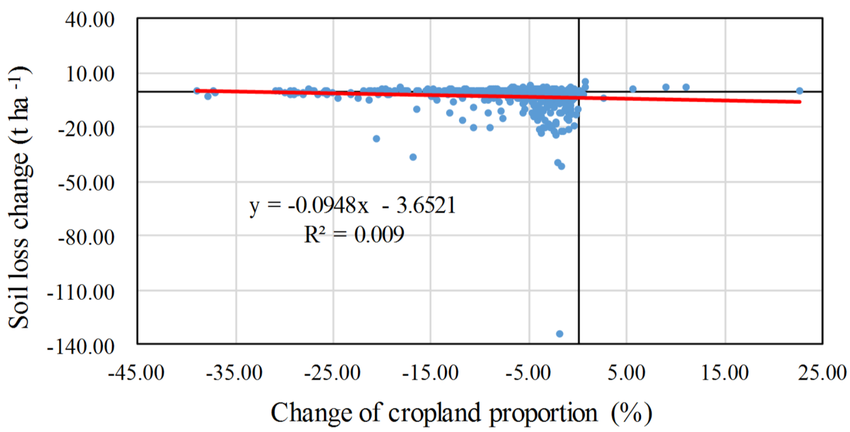

Fig. S4. Relationship between change of cropland proportion and soil loss change.

Table S1

The main data sources.

Data	Resolution	Source
Digital elevation model	90m	http://hydrosheds.org/
Precipitation	Daily (23 stations)	http://data.cma.cn/
Soil map	1:1,000,000	The Second Soil Survey in China
Land use map	30m	Zhejiang ecosystem assessment data
Vegetation coverage map	250m	Zhejiang ecosystem assessment data

Table S2

Observation data on sediment yield, SDR and soil loss in the Qiantang River Basin.

	Reference	Site (measured area)	Time	Observations
Sediment yield	MWR[1]	Lanxi (1.82)	1977-2015	1.24 t/ha
		Quzhou (0.54)	1958-2015	1.91t/ha
		Zhuji (0.17)	1956-2015	0.98t/ha
		Shangyu (0.45)	2012-2015	1.05t/ha
	Zhang et al.	Lanxi	1960-2012	242.38*10^4t
			1989-2000	206.00*10^4 t
			2000-2009	118.46*10^4 t
SDR	MWR[2]	Qiantang (5.56)		0.11
	Li et al.	Lanxi		0.132
Soil loss	Li et al.	Qiantang		4688.73*10^4t
	MWR[2]	Shuangyu	1958-2000	17.00 t/ha
		Lanxi	1977-2000	12.00 t/ha
		Zhuji	1958-2000	10.10 t/ha

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MWR[1]:

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Table S3

The proportions of land use, the values of soil loss, SDR and sediment export and changes in these values from 1990 to 2015 in the study area.

	1990	2000	2010	2015	1990-2000	2000-2010	2010-2015	1990-2015
Forestland (%)	59.63	59.88	61.91	62.38	0.25	2.03	0.47	2.75
Shrubland (%)	2.66	2.66	2.15	1.92	0.00	-0.51	-0.23	-0.74
Grassland (%)	2.83	2.49	1.27	1.06	-0.34	-1.22	-0.21	-1.77
Water (%)	5.52	5.50	5.39	5.26	-0.02	-0.11	-0.13	-0.26
Paddy field (%)	16.38	14.90	13.15	12.18	-1.48	-1.75	-0.97	-4.20
Dry land (%)	7.25	6.72	6.30	5.94	-0.53	-0.42	-0.36	-1.31
Garden plot (%)	1.45	1.81	1.21	1.14	0.36	-0.60	-0.07	-0.31
Urban land (%)	4.01	5.77	8.58	10.08	1.76	2.81	1.50	6.07
Bare land (%)	0.28	0.26	0.05	0.03	-0.02	-0.21	-0.02	-0.25
Change area (%)	-	-	-	-	3.01	8.01	2.33	12.50
SL (t ha-1 yr-1)	13.79	13.77	10.88	10.70	-0.02	-2.89	-0.18	-3.09
SDR	0.1040	0.1031	0.1011	0.1003	-0.0009	-0.0020	-0.0008	-0.0037
SE (t ha-1 yr-1)	1.69	1.69	1.26	1.22	0.00	-0.43	-0.04	-0.47

Note: SL: soil loss; SDR: sediment delivery ratio; SE: sediment export.

Table S4

Land use conversion matrix from 1990 to 2015 (in km^2).

	2015									
1990	Forestland	Shrubland	Grassland	Water	Paddy Field	Dry land	Garden plot	Urban land	Bare land	Total
Forestland	25319.54	0.30	0.06	63.30	1.09	2.99	0.06	44.49	0.00	25431.83
Shrubland	276.07	801.82	0.06	5.87	0.29	0.41	0.31	48.80	0.04	1133.67
Grassland	185.29	2.29	440.47	3.50	112.67	241.64	5.44	215.78	0.07	1207.15
Water	4.15	0.27	0.49	2129.95	43.64	9.96	0.39	163.58	0.12	2352.55
Paddy Field	298.52	4.76	4.52	24.39	4754.63	383.55	126.13	1386.67	1.66	6984.83
Dry land	202.81	5.38	2.02	8.38	275.19	1857.44	35.22	706.82	0.18	3093.44
Garden plot	234.42	0.68	0.76	2.50	3.44	33.75	314.61	26.47	0.01	616.64
Urban land	4.83	0.12	0.24	5.69	5.39	3.00	1.13	1691.15	0.05	1711.60
Bare land	82.77	3.57	2.58	1.28	0.14	0.87	0.98	15.73	12.46	120.38
Total	26608.41	819.19	451.20	2244.86	5196.48	2533.61	484.27	4299.49	14.59	42652.09

Table S5

Sediment export change from different land use conversion from 1990 to 2015 (in t).

	2015								
1990	Forestland	Shrubland	Grassland	Water	Paddy field	Dry land	Garden plot	Urban land	Bare land
Forestland	-33459.83	9.02	5.51	-910.29	50.93	3852.64	62.32	283.99	10.40
Shrubland	-17841.11	-1897.07	5.15	-199.83	4.73	521.92	391.62	-1139.18	434.38
Grassland	-30366.20	-154.98	-1033.98	-154.21	-2637.08	111735.81	1869.26	-4003.67	323.51
Water	158.89	9.07	19.69	0.00	345.35	1934.15	347.37	916.92	0.35
Paddy field	-18958.19	-102.04	146.38	-423.03	-5218.11	172716.94	39958.70	-11085.24	4604.86
Dry land	-292483.68	-6230.22	-2136.09	-1327.81	-140993.24	-17787.59	-5874.53	-145570.75	313.19
Garden plot	-381033.47	-491.97	-480.64	-405.28	-3174.32	10574.82	-7443.58	-10526.74	15.88
Urban land	-44.10	1.68	7.84	-33.07	73.62	682.33	455.85	-577.72	52.99
Bare land	-1084218.79	-39844.60	-13057.97	-3526.92	-446.91	-1797.82	-8608.02	-40652.05	-1072.43