Particulate organic carbon dynamics with sediment transport in the upper Yangtze River

Yuyang Wu, Hongwei Fang, Lei Huang, Zhenghui Cui

PII: S0043-1354(20)30730-2
DOI: https://doi.org/10.1016/j.watres.2020.116193
Reference: WR 116193

To appear in: Water Research

Received date: 18 May 2020
Revised date: 14 July 2020
Accepted date: 15 July 2020

Please cite this article as: Yuyang Wu, Hongwei Fang, Lei Huang, Zhenghui Cui, Particulate organic carbon dynamics with sediment transport in the upper Yangtze River, Water Research (2020), doi: https://doi.org/10.1016/j.watres.2020.116193

This is a PDF file of an article that has undergone enhancements after acceptance, such as the addition of a cover page and metadata, and formatting for readability, but it is not yet the definitive version of record. This version will undergo additional copyediting, typesetting and review before it is published in its final form, but we are providing this version to give early visibility of the article. Please note that, during the production process, errors may be discovered which could affect the content, and all legal disclaimers that apply to the journal pertain.

© 2020 Published by Elsevier Ltd.
Highlights

- We model organic carbon adsorption on sediment in the dammed upper Yangtze River.
- Our model integrates sediment properties and environmental factors.
- Cascade dams reduce suspended sediment and increase particulate organic carbon.
- Dam impacts are pronounced locally and further upstream and downstream.
Particulate organic carbon dynamics with sediment transport in the upper Yangtze River

Yuyang Wu\textsuperscript{a}, Hongwei Fang\textsuperscript{*a}, Lei Huang\textsuperscript{*a}, Zhenghui Cui\textsuperscript{b}

\textsuperscript{a} State Key Laboratory of Hydro-science and Engineering, Department of Hydraulic Engineering, Tsinghua University, Beijing, 100084, China

\textsuperscript{b} China Renewable Energy Engineering Institute, Beijing, 100120, China

Abstract

Understanding the behaviour of particulate organic carbon (POC) with sediment transport allows for a more accurate estimation of global carbon cycling and the conditions of aquatic ecosystems. River damming alters POC dynamics profoundly by the retention of sediments on which organic carbon (OC) is adsorbed. In this study, we developed a mechanism-based approach to investigate organic carbon (OC) adsorption on river sediment, integrating sediment particle properties (particle size, particle density, surface site density, and particle morphology) and environmental factors (dissolved OC concentration, pH, and suspended sediment concentration). We used this approach to assess the POC concentration in the dammed upper Yangtze River and to compare it with observed POC values in literature; model results and observations correlated very well ($R^2 = 0.89$; NSE = 0.83; $p < 0.001$). OC adsorption on sediment was found to correlate positively with dissolved organic carbon concentration and negatively with pH and suspended sediment concentration. We found that hydroelectric cascade development contributed to a decrease in suspended sediment concentration, with a significant increase in POC concentration both at reservoir sites, and upstream and downstream. The average suspended sediment concentration near the watershed outlet decreased from 2.08 kg m$^{-3}$ (individual reservoir period) to 1.57 kg m$^{-3}$ (early stage of reservoir construction/operation) and then to 0.01 kg m$^{-3}$ (cascade reservoir period). In contrast, the average POC concentration in the dammed river increased from less than, or around 1\%, to 1\% and finally to 3\% during each of these three periods, respectively. Our results highlight the pronounced impacts of cascade reservoirs on river sediment and POC dynamics. By providing a method for assessing OC adsorption on sediment and the dynamics of...
POC in aquatic systems, this work advances our understanding of carbon cycling in aquatic systems in times of global change.

**Keywords:** particulate organic carbon; adsorption; soil and sediment properties; hydroelectric dam; carbon cycle; modelling

1. Introduction

The transport of organic carbon (OC) from terrestrial to aquatic systems is a key link in the global carbon cycle (Ludwig et al., 1996; Cole et al., 2007). Understanding OC transport and processes that affect OC fate is critical to the assessment of carbon flux from land to the coastal ocean (Canuel and Hardison, 2016). Based on filter pore size, OC is divided into two principal fractions: dissolved organic carbon (DOC: < 0.2/0.45/0.7 µm) and particulate organic carbon (POC: > 0.2/0.45/0.7 µm) (Szymczycha et al., 2017; Derrien et al., 2019). Prior research estimated that the OC flux to the global ocean is 300 TgC yr\(^{-1}\), with POC being 140 TgC yr\(^{-1}\) (Seitzinger et al., 2010). Sediment is the major carrier of POC in aquatic environments (Wu et al., 2018). The interface between land and ocean is heterogeneous due to diverse sediment transport dynamics and sedimentation conditions (Bianchi et al., 2018). The role of soil erosion and sediment deposition in the carbon balance, and whether soil and sediment carbon are sources of, or sinks for, atmospheric carbon, have been debated for a long time (Harden et al., 1999; Smith et al., 2001; Lal, 2003). In this context, a better insight into the processes of POC transport with sediment is required. The quantification of OC adsorption on sediment during surface runoff and river flow is key for assessing the carbon budget across the terrestrial–aquatic interface (Veyssy et al., 1998). Nonetheless, a comprehensive understanding of the POC dynamics during sediment transport from terrestrial to aquatic systems is still lacking.

Earlier workers have used linear, non-linear, Freundlich, and Langmuir isotherms, and their revised versions, to describe the adsorption of OC on soils and sediments (Moore et al., 1992; Kothawala et al., 2009). As these studies indicate, OC adsorption is influenced by the properties of soil and sediment particles, and, also, by environmental factors (Boithias et al., 2014). Qu et al. (2020) found that POC in sediment decreases with the sediment’s median particle size but increases with its specific surface area (SSA). Interactions between OC and mineral particles are difficult to generalise, because sediments can consist of various mineral grains of different physicochemical properties (LaRowe et al., 2020). Analysis of pure mineral components of natural soils and
sediments has been helpful in determining OC adsorption capacity (Jardine et al., 1989). Nowadays, however, the maximum adsorbed OC is mainly identified through batch experiments on soils and sediments (Vandenbruwane et al., 2007; Kothawala et al., 2009). In addition, much work has been done on the effects of environmental factors, such as DOC concentration, ionic strength, and pH, on the adsorption of OC on clay mineral surfaces (Shen, 1999; Perez et al., 2011). Recently, more attention has also been paid to the influence of suspended sediment concentration on pollutant adsorption in aquatic environments (Huang et al., 2017). When the total suspended solids (TSS) are very low, OC adsorption onto sediment is high, similar to the OC values in autochthonous pools, as a result of primary production (Ludwig et al., 1996). At higher TSS values, OC adsorption on sediment decreases, and may drop to values close to the local soil organic content (SOC) (Fabre et al., 2019). This may be because suspended sediment limits in situ OC production by reducing sunlight penetration in the water and by diluting POC (Ludwig et al., 1996). Understanding the interaction between OC and sediment particles is particularly important for predicting the behaviour of OC in aquatic systems.

OC is readily adsorbed onto the surface of fine mineral particles eroded from soils. Traditional estimates of watershed-scale OC transport to water due to soil erosion were mainly based on empirical equations linking the suspended sediment concentration, SOC, and the OC enrichment ratio of the eroded soil (Starr et al., 2000), or on the empirical relationship between the sediment yield and the limited measurements of actual POC (Oeurng et al., 2011; Strauch et al., 2018). However, the disintegration of soil aggregates introduces great uncertainty into the fate of soil OC during erosion (Yadav and Malanson, 2009). The amount of POC transported with sediment in surface runoff, therefore, cannot be accurately estimated through the combined assessment of suspended sediment concentration and OC enrichment. Furthermore, the total POC concentration at the watershed scale is also difficult to identify with accuracy due to the contribution of other OC inputs (e.g., leaves, riparian and aquatic plants) into water (Lee et al., 2019). To this end, modelling the adsorption of OC on sediment during erosion and river transport, rather than assessing POC through empirical methods alone, can provide better estimates of POC concentration in large rivers at the watershed scale.

Human activities, particularly hydroelectric exploitation in large rivers, have altered aquatic environments and ecological situations drastically in the past decades. Sediment retention in dams profoundly affects the suspended sediment load, sediment delivery, and POC concentration in rivers (Sadaoui et al., 2018; Bianchi et al., 2020). Damming increases the OC input from terrestrial vegetation and soil into water, thus influencing greenhouse gas emissions due to organic matter decomposition (Barros et al., 2011). Several studies address the problem of organic
matter flux in rivers and estuaries downstream of hydroelectric dams (Zhang et al., 2014; Sadaoui et al., 2018). The Yangtze River, in China, is one of the world’s largest rivers, with several giant dams in its upper reaches. In the Yangtze River, POC primarily originates from terrestrial sources (Cauwet and Mackenzie, 1993). In 2009, the sediment and POC load from the Yangtze River to the East China Sea were approximately 408 Mt yr\(^{-1}\) and 1.52 TgC yr\(^{-1}\), respectively (Wang et al., 2012). With the completion of hydroelectric reservoirs in Yangtze’s upper reaches, the river flow and sediment load changed (Fang et al., 2012). The Jinsha River, representative of the upper Yangtze River, has a length of 3,481 km, accounting for approximately 77% of the entire upper Yangtze River course. Cascade reservoirs were operational, under construction, or planned in the Jinsha River. Sediment retention in dam reservoirs is the main cause of the decrease in POC flux in China over the past 60 years (Liu et al., 2020). Variations in POC concentration, induced by hydroelectric exploitation, result in important perturbations in the carbon cycle. More work is needed to advance understanding of these phenomena.

Long-term POC variation in rivers was generally estimated by empirical models (Fabre et al., 2019; Liu et al., 2020). However, field measurement is rarely an easy task, and results from \textit{in situ} OC adsorption experiments cannot be readily extrapolated to the scale of large watersheds. Here, we present a novel approach for estimating the POC concentration in the dammed upper Yangtze River. This approach can contribute to the evaluation of the long-term POC dynamics in large watersheds. The main objectives of this work are to, (a) model the dynamics of OC adsorption on sediment particles through a mechanism-based approach; (b) determine the effects of sediment particle properties and environmental factors on OC adsorption; and, (c) quantify the POC concentration transported with suspended sediment in the dammed upper Yangtze River.

2. Methodology

2.1 Description of the study area

The Yangtze River, the third largest river in the world, with the sixth largest flow discharge and the fourth largest sediment load (Wang et al., 2018), is divided into three parts (Fig. 1). Its headwater, the Jinsha River, originates at the Geladandong (Snowy) Mountain on the Tibetan Plateau. The climate in the Jinsha River Basin varies greatly due to the variable elevation of the basin. The upstream basin has a typical highland climate with snowmelt events, while the middle and downstream basins are mainly characterised by a subtropical climate with an annual precipitation of around 1,000 mm. In Jinsha’s middle and lower reaches, the runoff is \(880 \times 10^8\) m\(^3\) yr\(^{-1}\).
(2006 to 2015) and the total sediment load is 85.6 Mt yr\(^{-1}\) (2002 to 2015), approximately (Zhang et al., 2019). As the main source of Yangtze’s sediment, the Jinsha River Basin contributes approximately 57\% of the sediment load in the upper Yangtze River (Li et al., 2018).

The Jinsha River contains abundant hydropower resources, with a potential capacity of about 83.2 million kW, and annual power generation of 362 billion kWh (Wu et al., 2020). More than 30 hydropower stations were operational, under construction, or planned in the Jinsha River trunk stream, and the Yalong River, its biggest tributary, in 2017. In the cascade of reservoirs, events occurring in the upstream reservoirs may significantly impact the events in the downstream ones (Cunha-Santino et al., 2017). Among these cascade reservoirs, the Ertan hydropower station has operated since 1998. Four giant hydropower stations (Wudongde, Baihetan, Xiluodu, and Xiangjiaba) with the total installed capacity of 46.5 million kW have been under construction or completed in the lower Jinsha River trunk stream since 2011. Cascade reservoirs account for about 45\% of the sediment load reduction in this river. Almost 90\% of sediment retention is due to the operation of the Xiluodu and Xiangjiaba hydropower stations since 2013 (Zhang et al., 2019). As a result, the POC load transported with sediment in the upper Yangtze River has been profoundly impacted by changes in river flow and sediment transport.

Fig. 1. Location and topography of the study area, and observation sites along the Yangtze River trunk stream (DEM: digital elevation model; SG: Shigu, PS: Pingshan, ZT: Zhutuo, CT: Cuntan, YC: Yichang, SS: Shashi, HK: Hankou, JJ: Jiujiang, DT: Datong, TGD: Three Gorges Dam).

2.2 Estimate of organic carbon adsorption

Organic carbon adsorption on sediment depends on sediment particle properties that determine the maximum OC adsorption capacity, and on environmental factors that influence how much OC can be adsorbed on sediment particles in aquatic environments. The equation used to estimate OC adsorption on sediment is, thus, as follows:

\[
q_e = q_{\text{max}} \cdot F(\text{Environmental factors}),
\]

where \(q_e\) is the OC adsorbed on sediment at equilibrium (mg g\(^{-1}\)), \(q_{\text{max}}\) is the OC adsorption capacity of the sediment (mg g\(^{-1}\)), and \(F(\text{Environmental factors})\) is a function of environmental factors. The \(q_{\text{max}}\) is determined by intrinsic properties of the sediment particles, including the particle size distribution, particle surface structure, and mineralogy (Qu et al., 2020). To describe the properties of the sediment particles, we used the comprehensive factor \(P\) (Fang et al., 2017):
\[ P(D, N_s, F_{2a}) = \frac{N_s}{\rho} \left( \frac{2}{D} \right)^{\text{LSy} F_{2a} + 0.60}, \quad (2) \]

where \( D \) is the median particle size (\( \mu \text{m} \)), \( N_s \) is the surface site density (site \( \text{nm}^{-2} \)) for the interaction (proton exchange) between the sediment particle and OC, \( F_{2a} \) is the weighted average descriptor for sediment particle morphology, and \( \rho \) is the particle density (g cm\(^{-3} \)).

The relationship between \( q_{\text{max}} \) and \( P \) is expressed with the following empirical formula that quantifies the effects of mineral sediment particles on maximum OC adsorption (Fang et al, 2017):

\[ q_{\text{max}} = f(P) = aP + b, \quad (3) \]

where \( a \) and \( b \) are empirical parameters. These parameters describe soil and sediment mineral properties.

The environmental factors affecting OC adsorption on sediment particles generally include the aqueous equilibrium concentration of OC (\( C_e \)), pH, ionic strength (\( I \)), and the concentration of suspended solids (TSS). Considering these environmental parameters as individual functions, their combined effect on OC adsorption on sediment particles is expressed as follows:

\[ F(\text{Environmental factors}) = f(C_e) \cdot f(\text{pH}) \cdot f(I) \cdot f(TSS), \quad (4) \]

where \( f(C_e) \), \( f(\text{pH}) \), \( f(I) \), and \( f(TSS) \) are the functions of \( C_e \), pH, \( I \), and TSS, respectively.

Moreover, the partition coefficient, \( K_d \), has been widely used to characterise the OC adsorption process in water–sediment–pollutant systems. Therefore, we also introduced this coefficient in our analysis (Nodvin et al., 1986):

\[ K_d = \frac{q_e}{C_e}. \quad (5) \]

### 2.3 Determination of adsorption capacity

To identify the inherent OC adsorption properties of natural sediment independently of site specificity and environmental influences, we used pure samples of the six minerals that make up the bulk of natural sediment in the Yangtze River basin: quartz, potassium feldspar, calcite, kaolin, montmorillonite, and hematite. These mineral samples were obtained from the National Centre of Reference Material (NCRM). The particle size of natural sediment was determined using a laser particle size analyser (HORIBA LA-920, Japan). The mineral composition of natural sediments was determined with an X-ray diffractometer (D/max_rA, Japan). The specific surface area
(SSA) was obtained through nitrogen gas isothermal adsorption and the Brunauer–Emmett–Teller (BET) method (Brunauer et al., 1938), using an ASAP 2020M system (Micromeritics, USA).

The weighted average morphological descriptor $F_{2a}$ describes the micromorphology (raised or sunken microrelief) of mineral particles, which significantly affect their surface charge distribution and sorption capacity. Changes in the micromorphology of mineral surfaces influence the specific affinity and local electrostatic forces that affect chemical complexation at reactive surface sites (Cui et al., 2017). Taking this into account, we recorded the micromorphology of mineral particles using a JEOLJSM-6310F scanning electron microscope (SEM), and processed the microscopic images. We used Matlab 2016 to extract the particles’ edge profile, and applied a mathematical characterisation method based on Taylor Series Expansion to describe the micro-morphology characteristics and calculate the morphological descriptor, $F_{2a}$. Details on the $F_{2a}$ calculation method are available in Cui et al. (2017). The particle surface site density ($N_s$) is affected by the particle’s mineral composition. $N_s$ values were obtained from previous studies, and were optimised based on OC adsorption data (White and Zelazny, 1988; Venema et al., 1998; Wieland and Stumm, 1992).

Natural sediments entering the Jinsha River originate from the erosion of soils composed of various minerals. Hence, the $F_{2a}$, $N_s$, and $\rho$ of natural sediment in the Jinsha River Basin was determined through the component additivity method (Davis et al., 1998). Calculations were based on the properties ($F_{2a}$, $N_s$, and $\rho$) and mass percentage of each mineral species.

The OC adsorption capacity of sediment transported from terrestrial to aquatic environments is often difficult to measure. Fine mineral particles can be significantly enriched in OC. It has been demonstrated that OC adsorbed on sediment (%) is very closely linked with the clay content of the suspended sediment (correlation coefficient $0.718$: Yu et al., 2011). Adsorption experiments on typical Yangtze River Basin soils further indicated that there is a linear relationship between the maximum adsorbed OC and clay content (Xu and Zhao, 2017). Therefore, the $q_{\text{max}}$ of sediments in the Jinsha River Basin can be estimated from its empirical linear relationship with the clay fraction. We estimated the $q_{\text{max}}$ based on measured POC values in the upper Yangtze River, which ranged from 20.78 % (when the reservoir water level reached 156 m in the Three Gorges Dam) to 0.3 %, which is close to the lowest soil organic carbon content in the watershed (Zhang et al., 2014; Wu et al., 2018).

A total of 19 main types of soils (felty soils, dark felty soils, and frigid calcic soils, etc. from which Jinsha River sediments are derived), occupying 98.62 % of the total Jinsha River watershed, were analysed to identify maximum OC adsorption. Our analysis focused on the erosion-prone top horizons of these soils (0-20 cm). Sample
selection was based on China’s Soil-Scientific Database and on field survey in the study area (Chen et al., 2005; Zhao et al., 2018a).

2.4 Determination of environmental factors

To quantify the impact of the OC aqueous equilibrium concentration on OC adsorption \( f(C_e) \), we developed an empirical equation based on the classic Freundlich and Langmuir isotherms and on previous work (Vandenbruwane et al., 2007):

\[
f(C_e) = k_e \cdot C_e^{m_1},
\]

where \( k_e \) and \( m_1 \) are empirical parameters calculated in previous experimental studies (Shen, 1999; Vandenbruwane et al., 2007; Kothawala et al., 2009; Xu and Zhao, 2017).

The impact of pH on OC adsorption \( f(\text{pH}) \) varies across different pH ranges: OC adsorption on sediment particles generally increases with increasing pH when pH is lower than 4.5-5, and decreases with increasing pH when pH ranges from 5 to 10 (Jardine et al., 1989; Shen, 1999). Langmuir isotherm calculation of OC adsorption as a function of pH (Pagnanelli et al., 2003) showed that the power function describes the pH impact better than the linear and exponential functions. Considering the pH range in natural water (pH > 4.5), and the quantitative relationships between OC adsorption and pH established from observed data (Huang et al., 2016; Pagnanelli et al., 2003), we proposed the following equation:

\[
f(\text{pH}) = k_{\text{pH}} \cdot \text{pH}^{m_2},
\]

where \( k_{\text{pH}} \) and \( m_2 \) are empirical parameters calculated in previous experimental studies (Jardine et al., 1989; Shen, 1999).

The influence of ionic strength on OC adsorption \( f(I) \) was weak at the ionic strength \( I = 0.001 \) to 0.1 mol L\(^{-1}\) using NaCl considered here (Jardine et al., 1989). NaCl of three concentrations (0.3, 0.03, and 0.003 mol L\(^{-1}\)) were used, and the final solution \( I \) were 0.1, 0.01, and 0.001 mol L\(^{-1}\), respectively (Jardine et al., 1989). Thus, we assumed that \( f(I) = 1 \). Finally, the influence of suspended sediment, \( f(\text{TSS}) \), was calculated from the following equation, based on field observation and adsorption experiments (Huang et al., 2017):

\[
f(\text{TSS}) = k_{\text{TSS}} \cdot \text{TSS}^{m_3},
\]

where \( k_{\text{TSS}} \) and \( m_3 \) were empirical parameters whose values are based on measured sediment concentration (Changjiang Water Resources Commission: CWRC), and POC measurements in the Yangtze River (Wu et al., 2007; Bao et al., 2014; Wu et al., 2018).
The following equation clarifies the influence of environmental factors on OC adsorption:

$$\frac{K_d}{q_{\text{max}}} = F(\text{Environmental factors}) = k \cdot C_e^{m_e} \cdot \text{pH}^{m_p} \cdot TSS^{m_s},$$

(9)

where $k$ is a dimensionless parameter, of value to be determined. A summary of variable names and definitions used in this study is shown in Table 1.

### Table 1. Variable names and definitions used in this study.

#### 2.5 Observed sediment and POC concentrations in the upper Yangtze River

The POC concentration in the upper Yangtze River was estimated by Equation 1. Daily flow discharge and suspended sediment concentration data for the 2007-2016 period, at all 10 hydrological stations (Shigu, Panzhihua, Luning, Tongzillin, Huatan, Pingshan, Jinanqiao, Wudongde, Baihetan, and Xiangjiaba) in the Jinsha River Basin (Fig. 1) were obtained from the CWRC, China. OC aqueous equilibrium data ($C_e$) were calculated from measured dissolved organic carbon content in the upper Yangtze River (Shi et al., 2016; Wu et al., 2007). POC data were obtained by field sampling in the Yangtze River (Wu et al., 2007; Bao et al., 2014; Wu et al., 2018). Measured pH values in the dry season, the flood season, and the normal flow season included data from the lower stream of the Jinsha River (Xiluodu, Pingshan, Xiangjiaba, Riverine Park) for the 2002-2003 period (Three Gorges Dam Engineering Development Corporation of China), and pH measurements in the Yangtze River for the 2003-2006 period (Chen and Tu, 2008). Model performance was evaluated with the correlation coefficient $R$ and Nash-Sutcliffe efficiency (NSE). These two indicators have been widely used for performance evaluation of watershed hydrologic and water quality model (Moriaisi et al., 2015). Details of evaluation methods were described by Tang and Maggi (2018).

### 3. Results

#### 3.1 Adsorption capacity

The mineral properties that affect OC adsorption on natural sediments and soils in the Jinsha River Basin are listed in Table 2. Quartz and potassium feldspar are the main mineral components of natural sediments. By proportions of different minerals representing natural sediment, the weighted average morphological descriptor ($F_{\text{2a}}$) is 0.148; the surface site density ($N_s$) is 4.51 site nm$^{-2}$; and the particle density ($\rho$) is 2.66 g cm$^{-3}$. The properties $P$
and \( q_{\text{max}} \) of topsoil layers in the Jinsha River Basin are listed in Table 3. The median soil particle size in the Jinsha River watershed is hard to constrain. To reflect the sediment characteristics at the watershed scale and to reduce calculation errors, we estimated the representative median particle size from the textural properties of the top (most vulnerable to erosion) horizons of the 19 main soil types in the Jinsha River Basin (Table 3). We used Equation 2 to calculate the properties factor \( P \) of sediment originating from erosion of these 19 topsoil types. Applying the component additivity method (Davis et al., 1998), we calculated the average value of maximum OC adsorption on sediment (average \( q_{\text{max}} \)) in the Jinsha River Basin, at 72.05 mg g\(^{-1}\). The \( q_{\text{max}} \) ranged from 2.06 % to 20.78 % (% of OC in sediment by weight). A linear relationship analysis between \( P \) and \( q_{\text{max}} \) (Equation 3) showed that \( a \) and \( b \) are 291.1 and 1.01, respectively \((R^2 = 0.71, p < 0.001)\) (Table 3). This establishes a quantitative relationship between the sediment’s OC adsorption capacity and its properties (median particle size, surface site density; morphological descriptor; and particle density), comprehensively described by \( P \). Our results demonstrate that the comprehensive factor, \( P \), can indeed describe a sediment’s OC adsorption capacity.

Table 2. Mineral properties of natural soils and sediments.

Table 3. Topsoil properties and maximum adsorption of OC on sediment, in the Jinsha River Basin.

3.2 Environmental factors

First, we quantified the effect of the OC aqueous equilibrium concentration \((C_e)\) on OC adsorption. The extensive literature on OC adsorption on mineral particles in soils with low humus/(other) organic content, and soils sharing similar characteristics with natural sediments in the Jinsha River Basin was consulted. We used a wide range of \( C_e \) values (Fig. 2), obtained from previous experimental studies (Shen, 1999; Vandenbruwane et al., 2007; Kothawala et al., 2009; Xu and Zhao, 2017), and Equation 9 (the relationship between \( K_d/q_{\text{max}} \) and \( C_e \)), to obtain the parameter \( m_1 \) (Fig. 2). As shown in Fig. 2, there is a close correlation between \( K_d/q_{\text{max}} \) and \( C_e \) \((R^2 = 0.82)\). Since \( m_1 - 1 = -0.934 \), we obtained \( m_1 = 0.066 \).

Fig. 2. Relationship between \( K_d/q_{\text{max}} \) and \( C_e \) (solid: regression line; dotted: 50 % confidence interval; dotted-dashed: 70 % confidence interval; dashed: 90 % confidence interval).
We then quantified the effects of pH on OC adsorption. Because the previous experiments (Vandenbruwane et al., 2007; Xu and Zhao, 2017) did not include a wide range of pH conditions, the relationship between $K_d(q_{\text{max}}; C_e^{m-1})$ and pH cannot be used for the identification of $m$. There are few experimental studies that consider both the $C_e$ and pH range in controlled experiments, therefore, we used data from the literature (Jardine et al., 1989; Shen, 1999) to quantify the impact of pH on OC adsorption (Fig. 3). These earlier results demonstrate that the OC adsorption on sediment particles generally increases with pH increasing from 3 to 4.5, but it decreases rapidly with further increase in pH above the value of 4.5. To some extent, this variable effect of pH can also be seen in the relationship between $K_d(q_{\text{max}}; C_e^{m-1})$ and pH (Fig. 3), notwithstanding the narrow range of pH values in earlier experiments. This pattern was also observed for pure minerals (Davis and Glour, 1981). Considering the pH range in natural river water (pH > 4.5), we fitted a power function ($y = A \cdot x^B$) in the adsorbed OC and pH values, for pH > 4.5 (Fig. 3); $B$ ranged from -1.80 to -1.19 (Table 4); therefore, average $m = -1.518$.

![Fig. 3. Effects of pH on OC adsorption on sediment particles (red dots indicate the data shown in Fig. 2).](image)

Table 4. Relationship between the organic carbon/matter adsorbed on sediment particles and pH (pH > 4.5).

The parameter $m$ and $k_{s0}$ for the upper Yangtze River were calculated from the empirical relationship between suspended sediment concentration and OC adsorption proposed by Huang et al. (2017):

$$q_{\text{e0}} = k_{s0} \cdot \text{TSS}^{m_3},$$

(10)

where $q_{\text{e0}}$ is the OC adsorption on sediment particles (mg g$^{-1}$), $k_{s0}$ is the empirical parameter. By inserting measured sediment concentration and POC values in the upper Yangtze River into Equation 10, we found that $k_{s0} = 2.858$ and $m_3 = -0.610$, based on calibration and validation results (Fig. S1).

The OC adsorbed on sediment particles $q_e$ is expressed as follows:

$$q_e = q_{\text{max}} \cdot k \cdot C_e^{m_3} \cdot \text{pH}^{m_5} \cdot \text{TSS}^{m_0}$$

$$= 72.05 \cdot k \cdot C_e^{0.066} \cdot \text{pH}^{-1.518} \cdot \text{TSS}^{-0.610}.$$  

(11)

From Equations 10 and 11, we obtain $k = 0.879$ on average. The POC concentration in the upper Yangtze River is thus estimated as follows:

$$q_e = 63.34 \cdot C_e^{0.066} \cdot \text{pH}^{1.518} \cdot \text{TSS}^{-0.610}.$$  

(12)
3.3 Comparison between observations and calculations

Owing to the geographical location and economic development of the Jinsha River Basin, extensive observed POC, DOC, pH, and suspended sediment concentration data were relatively unavailable. Observed data are also available for the middle and lower reaches of the Yangtze River. Based on the measured values of DOC, pH, and suspended sediment concentration, the POC concentration for the entire Yangtze River was calculated, and then compared with the measured POC. Validation analysis demonstrated a very good model performance based on the \( R^2 \) and NSE (\( R^2 = 0.89; \) NSE = 0.83; \( p < 0.001 \)) in the upper Yangtze River (Fig. 4), according to the evaluation criteria at the watershed scale suggested by Moriasi et al (2015). The R between the calculated and observed POC for the entire Yangtze River was 0.78 (\( p < 0.001 \)). Uncertainty analysis of POC estimate under empirical parameter variations was conducted. POC estimates under different parameters were compared with the POC estimate under the identified parameters in Equation 12. Only one parameter was changed, and the other ones were kept constant at a time. Parameters \( m_1 \), \( m_2 \), \( m_3 \), and \( k \) were changed by −20 % to +20 % with an increment of 5 %. Analysis showed that variations in \( m_2 \) caused the greatest uncertainty (Fig. S2). Model uncertainty was very low when parameter varied by −5 % to +5 % (0.87 < NSE < 0.99). This analysis validates our POC calculation approach for the upper Yangtze River.

Fig. 4. Comparison between the calculated and the observed POC in the Yangtze River (shaded areas from light to dark orange colour indicate the predictive intervals with 95 %, 75 %, and 50 % certainties, respectively; CT: Cuntan; SG: Shigu; ZT: Zhutuo; PS: Pingshan; YC: Yichang).

3.4 Variations in suspended sediment concentration

The suspended sediment concentration in the upper Yangtze River decrease markedly from the period of individual reservoir operation (2007-2011) to the early stage of cascade reservoir construction/operation (2012), and further when the cascade reservoir became operational (2015-2016) (Fig. 5). During 2007-2011, the highest suspended sediment concentration was measured at the Pingshan station (peak value: 2.08 kg m\(^{-3}\)); the second highest was at the Huatan (1.82 kg m\(^{-3}\)) (Fig. 5). In 2012, the suspended sediment concentration was high at the Shigu, Luning (tributary), Huatan, and Xiangjiaba stations; nevertheless, the suspended sediment concentration in
the lower stream was not higher than that in the upper stream (Fig. 5.B). The suspended sediment concentration at Xianjiaba peaked in September 2012 (1.57 kg m\(^{-3}\)). During 2015-2016, when the Xiluodu and Xiangjiaba hydropower stations had become operational, the suspended sediment concentration at Xiangjiaba was very low (average: 0.01 kg m\(^{-3}\)). The Shigu and Luning stations, which were less affected by the cascade reservoirs, had high suspended sediment concentrations: 1.45 kg m\(^{-3}\) in July, and 1.08 kg m\(^{-3}\) in June, on average. Moreover, the Panzhihua station, although at some distance from the cascade reservoirs, also showed a clear decrease in suspended sediment concentration: from 0.48 kg m\(^{-3}\) before reservoir construction (2007-2011), to 0.27 kg m\(^{-3}\) in the early stage of reservoir construction/operation; then to 0.07 kg m\(^{-3}\) during the operation of the cascade reservoir (average sediment April-November values). Analysis of variance (ANOVA) showed that the monthly variance between these three periods was significant in Panzhihua and Pingshan/Xiangjiaba. By retaining sediment, cascade reservoirs had a pronounced impact on the Yangtze River system.

Fig. 5. Variations in sediment concentration in the upper Yangtze River trunk stream during, (A) operation of individual reservoirs (2007-2011); (B) early stage of cascade reservoir construction/operation (2012); (C) cascade reservoir construction/operation (2015-2016) (Gray areas: flood season; SG: Shigu; PZH: Panzhihua; LN: Luning; TZL: Tongzilin; HT: Huatan; PS: Pingshan; JAO: Jianqiao; WDD: Wudongde; BHT: Baihetan; XJB: Xiangjiaba).

3.5 Variations in POC concentration

The effects of dam construction on POC concentration (% of POC in TSS) in the dammed upper Yangtze River were especially pronounced during the period of cascade reservoir construction and operation (Fig. 6). A marked increase in POC (%) was recorded at the Tongzilin, Xiangjiaba, and Jianqiao hydrological stations, located near the dam site. During the period of individual reservoir operation (2007-2011), the highest POC (3 %, in December) was at Tongzilin, 18 km from the Ertan hydropower station. In the middle to lower Jinsha River, the POC was less than, or around, 1 %. At the early stage of cascade reservoir construction/operation (2012), the highest POC was recorded at Xiangjiaba (5 ± 1 %), and the second highest at Tongzilin (4 %). The average POC in the middle to lower Jinsha River during this period was 1 %. During the main period of cascade reservoir construction/operation (2015-2016), the POC at Xiangjiaba (average concentration: 8 ± 1 %) was far higher than that in any other area. The POC at Jianqiao was also high, especially in November and December (average concentration: 7 ± 1 %). The average POC in the middle to lower Jinsha River during this period was 3 %, showing
that the cascade reservoir operation profoundly altered the OC dynamics in the river system. Analysis of variance (ANOVA) showed that the monthly variance in POC between these three periods was significant in Panzhihua, Luning, Huatan/Baihetan, and Pingshan/Xiangjiaba, which indicated that these areas were most affected by the cascade reservoirs.

Fig. 6. Variations in POC concentration in the upper Yangtze River trunk stream during, (A) operation of individual reservoirs (2007-2011); (B) early stage of cascade reservoir construction/operation (2012); (C) cascade reservoir construction/operation (2015-2016) (Gray areas: flood season; SG: Shigu; PZH: Panzhihua; LN: Luning; TZL: Tongzilin; HT: Huatan; PS: Pingshan; JAQ: Jinanqiao; WDD: Wudongde; BHT: Baihetan; XJB: Xiangjiaba).

4. Discussion

4.1 Effects of sediment properties on OC adsorption

The OC adsorption capacity and the properties factor (P) of the 19 main types of soils in the Jinsha River Basin exhibit a range of values. Topsoil erosion generally contributed the bulk of sediment input among surface and sub-surface materials to riverine sediment (Blair et al., 2003; Adams et al., 2015). The maximum OC adsorption on Jinsha River sediment (q\text{max}) at the watershed scale, calculated with an empirical method, based on the additivity of observed components, falls within the range of global values (Ludwig et al., 1996). Previous empirical studies have also suggested that there is a relationship between the clay fraction and the sediment’s OC adsorption capacity (Shen, 1999; Oren and Chefetz, 2012). Uncertainties notwithstanding, our results indicate that the method presented here is effective for estimating the OC adsorption capacity. The principal soils in the Jinsha River Basin (felty, dark felty, and frigid calcic soils) are all characterised by a low clay content in their top horizons, implying that sediments resulting from erosion of these soils may have an OC adsorption capacity lower than the maximum OC adsorption capacity observed over the entire Yangtze River.

The effects of sediment properties on OC adsorption capacity can be expressed by the comprehensive factor, P, which integrates median particle size, surface site density, a morphological descriptor, and particle density. Scanning electron microscopy can provide better insights to the mechanisms of OC adsorption on sediment, since changes in sediment type may occur at sub-centimetre scale (LaRowe et al., 2020). The linear fitting between P and q\text{max} was good, highlighting the role of all the above factors in OC adsorption on sediment. This also implied that sediment properties that determine the OC adsorption capacity are critical to POC estimates. Sediment particle size
and surface site density, are indeed known to exercise important effects on OC adsorption dynamics during sediment transport (Haering et al., 2013; Bouchez et al., 2014; Fang et al., 2017). However, the impacts of mineral surface morphology on OC adsorption dynamics were seldom explored up to now. The OC adsorption capacity of sediments may differ even when their median particle sizes are the same (as also observed by Huang et al. (2016)). Understanding how changes in the sediment properties, alone or in combination, affect OC adsorption capacity will improve the evaluating of carbon dynamics in aquatic environments.

4.2 Effects of environmental factors on OC adsorption

In this study, we modelled the effects of the OC aqueous equilibrium concentration and pH on OC adsorption on sediments. The influence of environmental factors at the water–sediment interface on OC adsorption was extensively explored in earlier studies (Jin et al., 2008). This study went further than previous studies (Vandenbruwane et al., 2007; Xu and Zhao, 2017) to explore the combined effects of various environmental factors on OC adsorption dynamics. It is known that the effect of pH varies within different pH ranges, and that decrease in pH may contribute to the dissolution of OC when pH is less than 5 (Perez et al., 2011). In this study, we focused on the natural river environment, with pH higher than 4.5, and concluded that the OC adsorbed on sediments decreases as pH increased. The suspended sediment concentration, which varies widely in aquatic systems, was found to be the dominant factor controlling OC adsorption on sediment particles: the POC (%) tended to decrease with increasing concentration of suspended sediment. This varying pattern was also demonstrated by Ludwig et al. (1996) at the global scale.

Earlier studies estimated the POC at the watershed scale through empirical methods, by considering only soil organic carbon and suspended sediment concentration (Oeurng et al., 2011; Boithias et al., 2014; Fabre et al., 2019). However, if the multiple factors influencing POC are not taken into account in POC calculations, empirical methods may underestimate the POC by a wide margin (Veyssy et al., 1998). The multifactor approach that integrates sediment properties and environmental factors proposed here, therefore, greatly advances POC estimates in aquatic systems at the watershed scale. However, this model under predicted POC at higher POC concentrations (lower TSS concentrations in CT; Fig. 4). This may be because of the variations in TSS dynamics (Maggi and Tang, 2015) and the activities of phytoplankton and microorganisms. Therefore, this model needs improvement by considering dynamics of suspended matter (e.g. sinking, flocculation, and resuspension), and more environmental and biological-ecological factors. Advective flow of adsorbed OC can be enhanced due to the increasing exposure
of suspended matter dynamics to anthropogenic nutrient input (Tang and Maggi, 2016). In addition, warming temperatures generally lead to increased OC degradation (Gudasz et al., 2010). This increase is closely related to aquatic primary production and microbial activities (Sunagawa et al., 2015; Rantala et al., 2016). Living organisms (e.g. algae, microzooplankton) (Drummond et al., 2014; Tang and Maggi, 2018) play a critical role in OC fate. Optimisation of the model developed here is the priority for future research, which will make a significant contribution to the literature on carbon cycling in aquatic systems.

4.3 Impacts of hydroelectric exploitation on the POC concentration

The impact of cascade reservoirs on the reduction of suspended sediment in the Jinsha River is pronounced. Although soil and water conservation may account for approximately 35% of the total reduction in suspended sediment concentration (Zhang et al., 2019), cascade hydroelectric exploitation was the dominant reason for this reduction. The impacts of hydroelectric exploitation on the aquatic environment at a regional, or even global scale, are not fully understood (Vorosmarty et al., 2003). In the Jinsha River Basin, reduction in suspended sediment concentration occurred around reservoirs and their nearby areas (e.g. Panzhihua) and, also, further downstream, at the lower Jinsha River and its tributary. Reservoirs propagate their effect upstream and downstream: by constructing the movement and exchange of sediments, organic matter and nutrients, they impact river biodiversity and the riverine ecosystem (Grill et al., 2019). Furthermore, cascade reservoirs may contribute to discontinuous fluctuations in sediment properties (Guo et al., 2020), thereby altering riverine OC dynamics. The impacts of hydroelectric exploitation – especially cascade dams – on the aquatic environment, therefore, warrant deeper investigation.

Cascade reservoirs lead to an increase in POC within TSS locally, around the reservoirs. Our results highlighted that hydroelectric exploitation has greatly altered the POC dynamics in the upper Yangtze River. The POC (%) during the period of cascade reservoir operation was five times higher than that during the period of individual reservoir operation. The variation in the POC displayed seasonal characteristics, with an increase in the dry period and a decrease in the flood period, confirming earlier findings (Zhang et al., 2012). The pH in the river water fluctuated seasonally from about 7.8 to 8.0, but this fluctuation impacted the POC concentration only slightly. Cascade reservoirs alter the shift between the dissolved and particulate phase of OC, and contribute to the decrease in POC fluxes in aquatic environments. Globally, cascade reservoirs are predicted to decrease the total POC delivery to the ocean by 19% by 2030 (Maavara et al., 2017). This can have far-reaching impacts on the carbon
cycling from rivers to the ocean system, which have also been observed in other Asian rivers, such as the Yellow River (Xia et al., 2016), the Mekong River (Zhao et al., 2018b), and the Red River (Thi et al., 2017). Since the POC-TSS dynamics are closely related to hydrological and ecological processes in the aquatic environment (Derrien et al., 2019), the investigation of the impacts of hydroelectric exploitation on aquatic systems calls for a holistic approach at the watershed scale (Wu et al., 2020).

5. Conclusions

A mechanism-based approach, integrating sediment particle properties (particle size, particle density, surface site density, morphological descriptor) and environmental factors (dissolved organic carbon concentration, pH, suspended sediment concentration), was developed to model the dynamics of organic carbon adsorption on river sediment. This approach produced reliable estimates of the particulate organic carbon concentration in the dammed upper Yangtze River. The observed and calculated particulate organic carbon concentrations correlated very well ($R^2 = 0.89$; NSE = 0.83; $p < 0.001$), demonstrating the reliability of the proposed model. This model enabled us to explore the dynamics of particulate organic carbon transported with sediment in the upper Yangtze River. Cascade reservoirs led to a decrease in suspended sediment concentration, and a significant increase in particulate organic carbon concentration in local areas around the reservoirs and the watershed outlet. The average particulate organic carbon (% in TSS) increased from less than, or around 1% during the period of individual reservoir operation to 1% during the early stage of reservoir construction/operation, and, finally, to 3% during the period of cascade reservoir operation. The impacts of cascade reservoirs on suspended sediment concentration and particulate organic carbon were pronounced in both the reservoir areas and further upstream and downstream. The method for estimating particulate organic carbon proposed here can advance research into carbon cycling in aquatic systems in times of global change. Future work should focus on the optimisation of the model by considering dynamics of suspended matter and more environmental and biological-ecological factors.

Acknowledgements

We thank the editor and reviewers for their very helpful comments and suggestions. This work was supported by the National Natural Science Foundation of China (No. 91647210), 111 Project (No. B18031), and China Postdoctoral Science Foundation (No. 2018M641374).
Declaration of interests

☒ The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

☐ The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

References


Zhao, Y., Duan, X., Han, J., Yang, K., Xue, Y., 2018a. The main influencing factors of soil mechanical characteristics of the gravity erosion environment in the dry-hot valley of Jinsha river. Open Chem. 16, 796-809.

**Figures**

![Map of study area](image_url)

Fig. 1. Location and topography of the study area, and observation sites along the Yangtze River trunk stream (DEM: digital elevation model; SG: Shigu, PS: Pingshan, ZT: Zhutuo, CT: Cuntan, YC: Yichang, SS: Shashi, HK: Hankou, JJ: Jiujiang, DT: Datong, TGD: Three Gorges Dam).
Fig. 2. Relationship between $K_d/q_{\text{max}}$ and $C_e$ (solid: regression line; dotted: 50 % confidence interval; dotted-dashed: 70 % confidence interval; dashed: 90 % confidence interval).

Fig. 3. Effects of pH on OC adsorption on sediment particles (red dots indicate the data shown in Fig. 2).
Fig. 4. Comparison between the calculated and the observed POC in the Yangtze River (shaded areas from light to dark orange colour indicate the predictive intervals with 95 %, 75 %, and 50 % certainties, respectively; CT: Cuntan; SG: Shigu; ZT: Zhutuo; PS: Pingshan; YC: Yichang).
Fig. 5. Variations in sediment concentration in the upper Yangtze River trunk stream during, (A) operation of individual reservoirs (2007-2011); (B) early stage of cascade reservoir construction/operation (2012); (C) cascade reservoir construction/operation (2015-2016) (Gray areas: flood season; SG: Shigu; PZH: Panzhihua; LN: Luning; TZL: Tongzilin; HT: Huatan; PS: Pingshan; JAQ: Jinanqiao; WDD: Wudongde; BHT: Baihetan; XJB: Xiangjiaba).
Fig. 6. Variations in POC concentration in the upper Yangtze River trunk stream during, (A) operation of individual reservoirs (2007-2011); (B) early stage of cascade reservoir construction/operation (2012); (C) cascade reservoir construction/operation (2015-2016) (Gray areas: flood season; SG: Shigu; PZH: Panzhihua; LN: Luning; TZL: Tongzilin; HT: Huatan; PS: Pingshan; JAQ: Jinanqiao; WDD: Wudongde; BHT: Baihetan; XJB: Xiangjiaba).
### Tables

Table 1. Variable names and definitions used in this study.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Definition</th>
</tr>
</thead>
<tbody>
<tr>
<td>$q_e$</td>
<td>OC adsorbed on sediment at equilibrium (mg g$^{-1}$)</td>
</tr>
<tr>
<td>$q_{max}$</td>
<td>OC adsorption capacity of the sediment (mg g$^{-1}$)</td>
</tr>
<tr>
<td>$F$ (Environmental factors)</td>
<td>Function of environmental factors</td>
</tr>
<tr>
<td>$P$</td>
<td>Comprehensive factor that describes the properties of the sediment particles</td>
</tr>
<tr>
<td>$D$</td>
<td>Median particle size ($\mu$m)</td>
</tr>
<tr>
<td>$N_s$</td>
<td>Surface site density (site nm$^2$) for the interaction (proton exchange) between the sediment particle and OC</td>
</tr>
<tr>
<td>$F_{2a}$</td>
<td>Weighted average descriptor for sediment particle morphology</td>
</tr>
<tr>
<td>$\rho$</td>
<td>Particle density (g cm$^{-3}$)</td>
</tr>
<tr>
<td>$a$</td>
<td>Empirical parameter in Equation 3</td>
</tr>
<tr>
<td>$b$</td>
<td>Empirical parameter in Equation 3</td>
</tr>
<tr>
<td>$C_e$</td>
<td>Aqueous equilibrium concentration of OC (mg L$^{-1}$)</td>
</tr>
<tr>
<td>$I$</td>
<td>Ionic strength (mol L$^{-1}$)</td>
</tr>
<tr>
<td>$TSS$</td>
<td>Total suspended solids (kg cm$^{-3}$)</td>
</tr>
<tr>
<td>$K_d$</td>
<td>Partition coefficient in Equation 5</td>
</tr>
<tr>
<td>$k_e$</td>
<td>Empirical parameter in Equation 6</td>
</tr>
<tr>
<td>$m_1$</td>
<td>Empirical parameter in Equation 6</td>
</tr>
<tr>
<td>$k_{pH}$</td>
<td>Empirical parameter in Equation 7</td>
</tr>
<tr>
<td>$m_2$</td>
<td>Empirical parameter in Equation 7</td>
</tr>
<tr>
<td>$k_{TSS}$</td>
<td>Empirical parameter in Equation 8</td>
</tr>
<tr>
<td>$m_3$</td>
<td>Empirical parameter in Equation 8</td>
</tr>
<tr>
<td>$k$</td>
<td>Empirical parameter in Equation 9</td>
</tr>
<tr>
<td>$q_{e0}$</td>
<td>OC adsorption on sediment particles (mg g$^{-1}$) in Equation 10</td>
</tr>
<tr>
<td>$k_{S0}$</td>
<td>Empirical parameter in Equation 10</td>
</tr>
</tbody>
</table>
Table 2. Mineral properties of natural soils and sediments.

<table>
<thead>
<tr>
<th>Mineral type</th>
<th>SSA (m² g⁻¹)</th>
<th>F₂α</th>
<th>Nₛ (site nm⁻²)</th>
<th>ρ (g cm⁻³)</th>
<th>Proportion (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Quartz</td>
<td>3.82</td>
<td>0.114 ± 0.015</td>
<td>4.80</td>
<td>2.65</td>
<td>53.7</td>
</tr>
<tr>
<td>Potassium feldspar</td>
<td>4.31</td>
<td>0.161 ± 0.016</td>
<td>5.35</td>
<td>5.10</td>
<td>34.2</td>
</tr>
<tr>
<td>Calcite</td>
<td>1.92</td>
<td>0.106 ± 0.012</td>
<td>4.25</td>
<td>2.65</td>
<td>5.60</td>
</tr>
<tr>
<td>Kaolin</td>
<td>16.94</td>
<td>0.444 ± 0.039</td>
<td>2.06</td>
<td>2.60</td>
<td>3.30</td>
</tr>
<tr>
<td>Montmorillonite</td>
<td>27.36</td>
<td>0.379 ± 0.028</td>
<td>3.63</td>
<td>2.57</td>
<td>2.70</td>
</tr>
<tr>
<td>Hematite</td>
<td>5.62</td>
<td>0.142 ± 0.056</td>
<td>5.00</td>
<td>2.70</td>
<td>0.50</td>
</tr>
<tr>
<td>Weighted average</td>
<td>4.96</td>
<td>0.148 ± 0.017</td>
<td>4.51</td>
<td>2.66</td>
<td>-</td>
</tr>
</tbody>
</table>
Table 3. Topsoil properties and maximum adsorption of OC on sediment, in the Jinsha River Basin.

<table>
<thead>
<tr>
<th>Soil type</th>
<th>Clay (%)</th>
<th>Silt (%)</th>
<th>Sand (%)</th>
<th>SOC (%)</th>
<th>P (%)</th>
<th>$q_{max}$ (%)</th>
<th>Proportion (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Yellow brown earth</td>
<td>55.80</td>
<td>37.66</td>
<td>25.24</td>
<td>3.69</td>
<td>0.468</td>
<td>20.78</td>
<td>5.98</td>
</tr>
<tr>
<td>Brown earth</td>
<td>21.30</td>
<td>37.00</td>
<td>41.70</td>
<td>3.90</td>
<td>0.244</td>
<td>8.12</td>
<td>5.66</td>
</tr>
<tr>
<td>Dark brown earth</td>
<td>15.40</td>
<td>48.93</td>
<td>35.67</td>
<td>5.83</td>
<td>0.265</td>
<td>5.95</td>
<td>5.51</td>
</tr>
<tr>
<td>Purple soil</td>
<td>16.00</td>
<td>19.50</td>
<td>64.50</td>
<td>1.17</td>
<td>0.099</td>
<td>6.17</td>
<td>6.02</td>
</tr>
<tr>
<td>Felty soil</td>
<td>12.00</td>
<td>37.55</td>
<td>50.45</td>
<td>1.09</td>
<td>0.180</td>
<td>4.70</td>
<td>23.29</td>
</tr>
<tr>
<td>Dark felty soil</td>
<td>18.60</td>
<td>52.65</td>
<td>28.75</td>
<td>3.03</td>
<td>0.309</td>
<td>7.13</td>
<td>11.05</td>
</tr>
<tr>
<td>Frigid calcic soil</td>
<td>12.30</td>
<td>46.55</td>
<td>41.15</td>
<td>3.05</td>
<td>0.247</td>
<td>4.81</td>
<td>11.58</td>
</tr>
<tr>
<td>Frigid frozen soil</td>
<td>8.50</td>
<td>13.31</td>
<td>78.19</td>
<td>3.06</td>
<td>0.079</td>
<td>3.42</td>
<td>5.69</td>
</tr>
<tr>
<td>Red earth</td>
<td>30.40</td>
<td>35.06</td>
<td>34.54</td>
<td>1.40</td>
<td>0.306</td>
<td>11.46</td>
<td>7.67</td>
</tr>
<tr>
<td>Boggy soil</td>
<td>13.70</td>
<td>39.56</td>
<td>46.74</td>
<td>6.46</td>
<td>0.194</td>
<td>5.33</td>
<td>3.76</td>
</tr>
<tr>
<td>Yellow earth</td>
<td>38.00</td>
<td>39.93</td>
<td>22.07</td>
<td>2.11</td>
<td>0.508</td>
<td>14.25</td>
<td>2.01</td>
</tr>
<tr>
<td>Cinnamon soil</td>
<td>21.50</td>
<td>39.73</td>
<td>38.77</td>
<td>1.79</td>
<td>0.258</td>
<td>8.19</td>
<td>1.59</td>
</tr>
<tr>
<td>Brown coniferous forest soil</td>
<td>11.67</td>
<td>32.51</td>
<td>55.82</td>
<td>9.63</td>
<td>0.170</td>
<td>4.58</td>
<td>1.59</td>
</tr>
<tr>
<td>Gray cinnamonic soil</td>
<td>6.00</td>
<td>31.46</td>
<td>62.54</td>
<td>3.17</td>
<td>0.120</td>
<td>2.50</td>
<td>1.55</td>
</tr>
<tr>
<td>Paddy soil</td>
<td>22.00</td>
<td>44.82</td>
<td>33.18</td>
<td>0.99</td>
<td>0.301</td>
<td>8.37</td>
<td>1.25</td>
</tr>
<tr>
<td>Skeletal soil</td>
<td>4.80</td>
<td>47.57</td>
<td>47.63</td>
<td>0.23</td>
<td>0.203</td>
<td>2.06</td>
<td>1.23</td>
</tr>
<tr>
<td>Rocky soil</td>
<td>15.14</td>
<td>25.32</td>
<td>59.54</td>
<td>0.58</td>
<td>0.137</td>
<td>5.86</td>
<td>1.09</td>
</tr>
<tr>
<td>Meadow soil</td>
<td>34.30</td>
<td>54.92</td>
<td>10.78</td>
<td>1.15</td>
<td>0.550</td>
<td>12.89</td>
<td>1.06</td>
</tr>
<tr>
<td>Limestone soil</td>
<td>33.20</td>
<td>42.02</td>
<td>24.78</td>
<td>1.95</td>
<td>0.417</td>
<td>12.49</td>
<td>1.04</td>
</tr>
</tbody>
</table>
Table 4. Relationship between the organic carbon/matter adsorbed on sediment particles and pH (pH > 4.5).

<table>
<thead>
<tr>
<th>Experiment setting</th>
<th>Equation</th>
<th>$R^2$</th>
<th>$p$</th>
</tr>
</thead>
<tbody>
<tr>
<td>$C_e$ (5 mg L$^{-1}$)</td>
<td>$y = 16839.85x^{-1.83}$</td>
<td>0.84</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>$C_e$ (10 mg L$^{-1}$)</td>
<td>$y = 12663.86x^{-1.40}$</td>
<td>0.93</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>$C_e$ (15 mg L$^{-1}$)</td>
<td>$y = 15464.68x^{-1.43}$</td>
<td>0.93</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>$C_e$ (20 mg L$^{-1}$)</td>
<td>$y = 12154.67x^{-1.19}$</td>
<td>0.96</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>WI soil</td>
<td>$y = 844.34x^{-1.46}$</td>
<td>0.90</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>PC soil</td>
<td>$y = 1589.61x^{-1.80}$</td>
<td>0.89</td>
<td>&lt; 0.001</td>
</tr>
</tbody>
</table>
GRAPHICAL ABSTRACT