

## Article (refereed) - postprint

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Ji, Ningning; Liu, Yong; Wang, Shengrui; Wu, Zhihao; Li, Hong. 2022.  
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**The definitive version was published in *Water Research*, 216, 118350.  
<https://doi.org/10.1016>**

The definitive version is available at <https://www.elsevier.com/>

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1 **Buffering effect of suspended particulate matter on**  
2 **Phosphorus cycling during transport from rivers to lakes**

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21 **Abstract**

22       How to maintain harmful algal blooms under phosphate-limitation is still an open  
23 question in mesotrophic/eutrophic lakes. Little evidence for the importance of  
24 suspended particulate matter (SPM) in mediating phosphorus cycling and contributing  
25 to eutrophication has been generated for aquatic ecosystems, especially in coupled  
26 river-lake systems. In this study, we examined phosphorus transport and redistribution  
27 in a river-lake system in the Lake Erhai basin by establishing the relations between  
28 phosphorus distribution and phosphorus sorption behavior on SPM, and predicted how  
29 quality and quantity changes in SPM might influence phosphorus cycling by laboratory  
30 experiments and modeling. During the wet seasons, TP pool shifted from being  
31 dominated by total dissolved phosphorus (TDP) in the Miju River and estuary regions  
32 ( $73\pm 5\%$ ) to being dominated by total particulate phosphorus (TPP) ( $74\pm 11\%$ ) in Lake  
33 Erhai. The detritus-SPM in the Miju River as a P-sink and phytoplankton-SPM in Lake  
34 Erhai as a P-source buffered TDP levels during the wet seasons, which attributed to P  
35 activity and phytoplankton-POC of SPM. Increasing SPM concentrations could  
36 enhance the P-buffering. When  $C_0 \leq 5 \mu\text{mol/L}$  and phytoplankton-SPM  $\geq 16 \text{ mg/L}$ , P  
37 release increased by 50%-300%; when  $C_0 \geq 5 \mu\text{mol/L}$  and detritus-SPM  $\geq 16 \text{ mg/L}$ , P  
38 removal could exceed 30%. This study highlights two distinctive roles of SPM in  
39 regulating P cycling during transport from rivers to lakes. Especially the phytoplankton-  
40 SPM to buffer phosphate-limitation during algal blooms should not be ignored, which  
41 could provide theoretical references for the mechanism of continued algal blooms in  
42 mesotrophic lakes.

43 **Keywords:** Particulate phosphorus; Eutrophication; Suspended particulate matter;  
44 Phosphorus partition; Buffering

## 45 **1. Introduction**

46 Phosphorus (P) is an essential element for organisms and limits primary  
47 productivity in most freshwater systems, and understanding P cycling in aquatic  
48 systems is important to prevent eutrophication (Conley et al., 2009). In many eutrophic  
49 lakes, dissolved inorganic P (DIP), the almost 100% bioavailability, gradually declines  
50 during the peak algal bloom period (Reinhard et al., 2017), even below the limit of  
51 detection (Björkman and Karl, 2003; Ji et al., 2017), and total particulate phosphorus  
52 (TPP) dominates TP pool, such as more than 60% in Lakes Erie, Huron, Taihu and  
53 Dianchi (Lin and Guo, 2016; Xu et al., 2010; Wang et al., 2015). Studies have also  
54 shown that more than 80% of the fluvial flux of P to lakes may be associated with  
55 suspended particulate matter (SPM) (Baker et al., 2014; Pan et al., 2013). SPM can act  
56 as a source or sink of P by adsorption/desorption or uptake/release to regulate the  
57 abundance, bioavailability, and biogeochemical cycling of P in freshwater  
58 environments (Cyr et al., 2009). However, due to the complicated component of SPM  
59 and the difficulty of obtaining in water environment, previous studies mainly focused  
60 on simulating adsorption/desorption process of P by sediments (Pan et al., 2013; Yi et  
61 al., 2017; Deborde et al., 2007). Little has been done to address a quantitative analysis  
62 about changes in quality and quantity of P-associated SPM during transport from rivers  
63 to lakes, which severely restricts our understanding of how SPM affects the cycling and

64 partition of P (Lin and Guo, 2016).

65 Adsorption processes and adsorption reversibility of SPM are important for  
66 regulating P levels especially when water conditions change. The adsorption capacity,  
67 which can be quantified by equilibrium adsorption constants, is determined by the  
68 chemical composition and concentration of SPM and adsorbate concentration (Pan et  
69 al., 2002). Equilibrium phosphorus concentration at zero sorption ( $EPC_0$ ), a parameter  
70 obtained from phosphate-particles adsorption experiments, has been used to provide  
71 quantitative information on whether particles remove or release P to calculate the P flux  
72 in the particle-water interface (Pan et al., 2013). P particle-reactivity in the particle-  
73 water interface can be defined as the partition coefficient ( $K_d$ ) with the ratio between  
74 the particulate and dissolved phases in aquatic ecosystems (Huang et al., 2017; Lin and  
75 Guo, 2016).  $K_d$  has been observed to decline with increasing SPM in both marine and  
76 freshwater environments, showing the “particle concentration effect” (O’Connor and  
77 Connolly, 1980). However, no studies to date have systematically integrated all these  
78 adsorption properties to determine the potential effect of SPM on P cycling at an entire  
79 river-lake system (Pan et al., 2013, Yi et al., 2017; Lin and Guo, 2016). Studies  
80 combining field investigations, laboratory experiments and modeling are still few and  
81 would lead to a better understanding of the cycling pathways of P species and SPM.

82 Lake Erhai, the second largest plateau freshwater lake in southwestern China, is  
83 one of the three most concerned lakes with better-quality in China and an important  
84 source of domestic and irrigation water in Dali, Yunnan Province (Tong et al., 2017).  
85 Because of the increases in anthropogenic discharges and agriculture non-point

86 pollution in recent years on the lake fringes, cyanobacterial blooms frequently occur  
87 from July to October of every year in the lake (Zhang et al., 2015). Especially, two  
88 massive cyanobacteria blooms occurred in September and October in 2013. It is in a  
89 critical period of transformation from mesotrophic status to eutrophication. Lake Erhai  
90 is potentially P-limited (Yu et al., 2014). Steep gradients in P species exist from  
91 inflowing rivers to the lake. As shown in the figure 1 in this study, average  
92 concentrations of DIP (phosphate) dramatically decreased from 8.2  $\mu\text{mol/L}$  in the  
93 upstream river to 0.2  $\mu\text{mol/L}$  in open Lake Erhai across the estuary. Especially during  
94 algal blooms, DIP concentration in Lake Erhai is often less than 0.1  $\mu\text{mol/L}$ . However,  
95 algal blooms often last until November ( $\text{Chl}a > 20 \mu\text{g/L}$ ). TPP pool can be almost 4  
96 times larger than TDP pool, and over 10 times larger than DIP pool (Ji et al., 2017). The  
97 algal blooms are attributed in part to an increase in the concentration of DIP from TPP  
98 transformation (Baker et al., 2014). Consequently, we hypothesise that SPM may help  
99 P cycling to maintain high algal biomass under DIP-limitation. The interesting problem  
100 is further intensified whether SPM is changing and would help or hinder the P cycling  
101 or have no effect from river to lake, which is essential for guiding water management  
102 and for coupled river–lake modeling in global P cycling (Pan et al., 2002; Yi et al.,  
103 2017).

104 Thus, we carried out a systematic investigation of the spatiotemporal variations of  
105 P species and SPM across the boundary from the Miju River to Lake Erhai (Fig.S1).  
106 The aims were to (1) determine the potential of SPM as a source or sink of P throughout  
107 the entire river-lake; (2) identify the factors that influence P-buffering ability of SPM;

108 (3) predict how changes in SPM concentration affect P level and  $K_d$  by laboratory  
109 experiments and modeling; and (4) propose implications of SPM for P cycling during  
110 transport from river to lake. A summary of these experiments, as well as their respective  
111 objectives, is presented in [Fig.S2](#). These results are important for understanding the  
112 nature and controls on P cycling in particle rich rivers-lakes system.

## 113 **2. Materials and methods**

### 114 **2.1. Study area**

115 Lake Erhai, Yunnan Province, China, is at an altitude of about 1970 m and is  
116 between 25.6° and 25.9° N and 100.1° and 100.3° E. The lake covers approximately  
117 250 km<sup>2</sup> and is fed by a watershed that extends across 2565 km<sup>2</sup> ([Fig.S1](#)). It is, on  
118 average, 10.6 m deep. The area has a subtropical monsoon climate, with annual average,  
119 average maximum, and average minimum temperatures of 15.1, 20, and 8.9 °C,  
120 respectively. The mean annual rainfall is 1100 mm, of which 90% falls during wet  
121 seasons from May to October, and the remaining months are dry seasons.  
122 Cyanobacterial blooms usually appear during the wet seasons, and even can persist until  
123 November. The relatively long residence time of water (average 2.75 years) could  
124 promote accumulation of phytoplankton biomass, nutrients and SPM in the lake, and  
125 allow time for particle-water interaction to occur ([Zhang et al., 2015](#)). There are 22  
126 main rivers inflowing into the lake as main pollutant sources. The water quality of the  
127 rivers is mostly ranked as class V ( $9.6 < TP \leq 12.9$   $\mu\text{mol/L}$ ), the worst category of  
128 Chinese Surface Water Quality Standards (GB3838-2002) ([Ji et al., 2017](#)). Of these

129 rivers, Miju, which flows into the northern part of the lake and is the greatest pollutant  
130 discharge (accounting for 57%) into the lake, is classified as worse than class V (Lu et  
131 al., 2015).

## 132 **2.2. Sampling and Analysis**

133 The 11 sampling sites were selected with GPS from the Miju River, through the  
134 river-lake mixing zone, into Lake Erhai over 4 occasions, in April, July, October of  
135 2013, and January 2014 (Fig. S1, Table S1). Water samples for the lake and the river  
136 were collected using a 5 L Plexiglas water sampler at depth of 0.5-1.0 m and 0.2-0.5 m  
137 below surface, respectively. Water temperature (WT), pH, and dissolved oxygen (DO)  
138 were measured *in situ* using portable YSI electrodes (Xylem Co. New York, USA). The  
139 flow was measured *in situ* with a flowmeter (Qliner2, HACH, USA). In the laboratory,  
140 SPM was obtained by filtering water samples using precombusted (550°C for 4 h) and  
141 preweighed glass-fiber filters (Whatman GF/F; nominal pore size 0.7  $\mu\text{m}$ ). The filters  
142 were then washed with deionized water and freeze-dried until a constant weight (Yang  
143 et al., 2021). Thus, dissolved P species here are operationally defined as those in the <  
144 0.7  $\mu\text{m}$  filtrate, which may contain living biomass and thus overestimate dissolved  
145 organic P (DOP) compared to those in the < 0.45  $\mu\text{m}$  filtrates. Water parameters,  
146 including TP, TDP, DIP and DOP, were analyzed according to Standard Methods  
147 (APHA 2012), details of which are given in Text S1.

## 148 **2.3. P speciation and organic matter in SPM**

149 SPM concentrations were calculated from the difference between the weights of  
150 the precombusted (550°C for 4 h) and weighed GF/F filters and the freeze-dried filter

151 samples per unit volume of water. TPP, particulate inorganic phosphorus (PIP), and  
152 particulate organic phosphorus (POP) were determined using the modified method of  
153 [Solórzano and Sharp \(1980\)](#), and particulate P fractions were analyzed using a  
154 sequential chemical extraction procedure ([Psenner and Pucsko, 1988](#)), details of which  
155 are given in [Text S2, Fig. S6](#). The five-step extraction sequence provides information  
156 about a range of fractions, namely (1) NH<sub>4</sub>Cl-P (2) BD-P (3) NaOH<sub>25</sub>-rP, NaOH<sub>25</sub>-nrP  
157 (4) HCl-P (5) NaOH<sub>85</sub>-P.

158 Particulate organic carbon (POC) and particulate organic nitrogen (PON) were  
159 measured with an element analyzer (Perkin-Elmer 2400 Series II) after vapor  
160 acidification to remove carbonates. Particulate organic matter (POM) was  
161 colorimetrically determined after treatment of the sample with potassium  
162 dichromate/H<sub>2</sub>SO<sub>4</sub> according to the Walkey–Black method ([Tan, 1995](#)). Particulate  
163 inorganic material (PIM) was then defined as the difference between SPM and POM.  
164 Chlorophyll- $\alpha$  (Chl $\alpha$ ) was determined spectrophotometrically after extraction with 90%  
165 acetone. The ratio between phytoplankton biomass (mg C /L) and POM (mg/L) was  
166 determined to quantify the phytoplankton carbon percentage of the POM  
167 (phytoplankton-POC) ([Sobczak et al., 2002](#)). Detrital origin POC (detritus-POC) was  
168 estimated as the difference between total POC and phytoplankton-POC. Gross primary  
169 productivity (GPP) was determined using the light and dark bottle oxygen technique  
170 ([Vollenweider, 1974](#)). Alkaline phosphatase activity (APA) was estimated using the  
171 model substrate p-nitrophenyl phosphate (pNPP, Sigma) ([Dore and Priscu, 2001](#)).  
172 Samples were analyzed in triplicate, and the standard deviations were always < 5%.

173 **2.4. Partition coefficient of P**

174 The partition of P between the dissolved and particulate phases and its particle  
175 reactivity can be quantified by the partition coefficient ( $K_d$ ) (Lin and Guo, 2016), which  
176 is calculated with the following equation:

177 
$$K_d = \frac{C_p}{C_d * [SPM]} \quad (1)$$

178 where  $C_p$  and  $C_d$  are the concentrations of particulate and dissolved P, respectively,  
179 in the inorganic, organic, or total P pools ( $\mu\text{mol/L}$ ).  $[SPM]$  is the concentration of SPM  
180 ( $\text{mg/L}$ ). Values of  $K_d$  are expressed in  $\text{L/mg}$ .

181 **2.5. Degree of P saturation**

182 The degree of P saturation (DPS), which relates a measure of P already adsorbed  
183 by SPM to its P adsorption capacity, could be a good indicator of P release potential  
184 from SPM to waters (Ohno et al., 2007). Acid ammonium oxalate extraction was used  
185 to determine DPS (McKeague and Day, 1966), details of which are given in Text S3.

186 We calculated the DPS with the following equation:

187 
$$DPS = \frac{P_{ox}}{0.5 * (Fe_{ox} + Al_{ox})} \quad (2)$$

188 Where  $P_{ox}$ ,  $Al_{ox}$ , and  $Fe_{ox}$  are expressed in  $\text{mg/g}$ .

189 **2.6. Adsorption Isotherms Experiments and Sorption Kinetics of P**

190 I: Experiments details of the adsorption isotherms and sorption kinetics of P are  
191 provided in the Text S4. Mathematical descriptions of the adsorption isotherms were  
192 described by a modified Freundlich equation as follows:  $Q_e = K_f (C_{eq}^n - EPC_0^n)$  (3)  
193 where  $Q_e$  is the amount of net sorption or release of P at equilibrium ( $\text{mg P /kg}$ ),  $K_f$  is  
194 the affinity constant ( $\text{L}/\mu\text{mol}$ ),  $n$  is the exponential factor, and  $C_{eq}$  is the equilibrium P

195 ( $\mu\text{mol/L}$ ).  $\text{EPC}_0$  is a measure of the P concentration at which SPM is neither adsorbing  
 196 nor releasing P in the equilibrium state ( $\mu\text{mol/L}$ ). Greater values of  $K_f$  and smaller  
 197 values of  $\text{EPC}_0$  indicate stronger P sorption ability of SPM (Jarvie et al., 2005). Model  
 198 parameters were estimated by a Marquardt nonlinear least-squares fitting routine.

199 In order to judge whether the SPM acts as a source or sink of P for the water body,  
 200 a criterion of  $\delta = C_{eq}^n - \text{EPC}_0^n$  (4) was defined (Pan et al., 2013)

201 When  $\delta < 0$ ,  $Q_e < 0$  (release), SPM is a source of P.

202 When  $\delta > 0$ ,  $Q_e > 0$  (adsorption), SPM is a sink for P.

203 The slope of the isotherm curve at  $Q_e=0$  is the distribution coefficient  $K_{d-eq}$ , which  
 204 is calculated by taking the derivative of the modified Freundlich equation with respect  
 205 to  $C_{eq}$  at the  $\text{EPC}_0$ .

$$206 \quad K_{d-eq} = d[K_f (C_{eq})^n] / d(C_{eq})_{\text{EPC}_0} = nK_f (C_{eq})^{n-1} \quad (5)$$

207 The removal rate of DIP can be described by the following equation:

$$208 \quad \text{DIP removal (\%)} = (C_0 - C_{eq}) / C_0 \quad (6)$$

209 Mathematical descriptions of the sorption kinetics were established by fitting the  
 210 data sets with first order sorption functions:  $Q_t = Q_{\max} (1 - e^{-kt})$  (7),  $Q_t$  (mg P /kg) is the  
 211 amount of sorbed P at time  $t$  (h),  $Q_{\max}$  (mg P /kg) is the amount of sorbed P in the  
 212 equilibrium.  $k$  is the first-order constant of sorption kinetics ( $\text{h}^{-1}$ ).

213 II: We conducted the adsorption experiment by changing the mixture mass ratio of  
 214 detritus-SPM and phytoplankton-SPM from 1:1 to 1:20 to simulate P adsorption  
 215 behavior of the increasing phytoplankton-SPM when algal blooms. The detritus-SPM  
 216 came from the station MR-1 in the Miju River in October, while phytoplankton-SPM

217 was obtained by adding fresh algae. After mixture, the remaining details of the  
 218 experiment are the same as described in Sorption Experiments I.

219 III: A series of SPM concentrations (2, 6, 10, 16, 25, 35 mg/L) were used to  
 220 determine the threshold of SPM to reduce or increase the DIP level by adsorption  
 221 experiments. These SPM concentrations were chosen to bracket the known changes of  
 222 SPM during different seasons over the past decades in the Miju River and Lake Erhai.  
 223 The phytoplankton-SPM and the detritus-SPM respectively came from the station LE-  
 224 2 in Lake Erhai due to its maximum phytoplankton-POC content and the station MR-1  
 225 in the Miju River due to its maximum PIM and detritus-POC content. Tubes were  
 226 spiked with  $\text{KH}_2\text{PO}_4$  to provide a range of initial P concentrations ( $C_0$ , 0.05–12  $\mu\text{mol/L}$ )  
 227 designed to simulate the typical condition of P input to the river and lake. The remaining  
 228 details of the experiment are the same as described in Sorption Experiments I.

### 229 **2.7. Modified $K_d$ model**

230 We modified the  $K_d$  model based on the equilibrium between the initial and final  
 231 TP concentrations used by [Prastka \(1998\)](#) and [Turner and Tyler \(1997\)](#) [eqn (8), (9)].  
 232 The modified  $K_d$  model can rationalize the apparent contradiction between the source  
 233 and sink of TDP on SPM under different conditions ([eqn (10)-(12)]). Refer to [Text S5](#)  
 234 for more details.

$$235 \quad \bar{A}_{\text{gpp}} = 19.8 \bar{TP} - 77 \quad (8)$$

$$236 \quad C_R + P_R + \frac{\bar{A}_1 + 77}{19.8} = C_{es} + P_{es} + \frac{\bar{A}_2 + 77}{19.8} \quad (9)$$

$$237 \quad K_d = \frac{P_{es}}{C_{es} * S_{es}} \quad (10)$$

$$238 \quad \Delta TDP = C_R - C_{es} = \frac{C_R K_d S_{es} - P_R + \frac{\bar{A}_2 - \bar{A}_1}{19.8}}{1 + K_d S_{es}} \quad (11)$$

239 
$$TDP\ removal\ (\%) = \frac{\Delta TDP}{C_R} \times 100 = \frac{K_d S_{es}}{1 + K_d S_{es}} - \frac{P_R}{C_R(1 + K_d S_{es})} + \frac{\bar{A}_2 - \bar{A}_1}{19.8 C_R(1 + K_d S_{es})} \quad (12)$$

240 where  $C_R$ ,  $P_R$  are the dissolved and particulate P concentrations in the initial  
 241 simulation, respectively, in mg/L;  $C_{es}$ ,  $P_{es}$ ,  $S_{es}$  are the final dissolved and particulate P  
 242 concentrations and SPM concentration, respectively, measured in mg/L;  $\bar{A}_{gpp}$  is the  
 243 average GPP, measured in g/m<sup>3</sup>/d;  $\bar{A}_1$ ,  $\bar{A}_2$  are the GPP in the initial and final  
 244 concentrations, respectively, in g/m<sup>3</sup>/d;  $K_d$  is the final partition of P species between the  
 245 dissolved and particulate phase for the steady state, in L/mg.

246 **2.8 Generalized additive model**

247 In this study, GAM was used to identify the relative influence of various factors  
 248 on P adsorption of SPM. Refer to [Text S6](#) for more details. The GAM model can be  
 249 described as below ([Zou et al., 2020](#)):

$$g(y) = s_0 + s_1(x_1) + \dots + s_m(x_m) + \varphi \quad (13)$$

250 where,  $s(x)$  represent smooth functions of linking explanatory variables;  $\varphi$  is the  
 251 random residual term;  $g(y)$  represent response variables.

252 **2.9. Statistical analysis**

253 Differences between treatments were determined using one-way analysis of  
 254 variance (ANOVA, SPSS, version 16.0). Tukey's multiple comparison test (honest  
 255 significant difference, HSD) was used to identify variances among groups ( $P \leq 0.05$ ).

## 256 **3. Results and Discussion**

### 257 **3.1. Variations in the species and partition of P during transport from rivers to** 258 **lakes**

259 Concentrations of different P species and SPM in the Miju river and Lake Erhai  
260 significantly varied between the dry and wet seasons (Fig. 1a). In the Miju River,  
261 concentrations of DIP, PIP and POP increased from  $2.7\pm 0.9$ ,  $0.3\pm 0.23$ ,  $0.5\pm 0.3$   $\mu\text{mol/L}$   
262 during the dry season to  $7.7\pm 2.8$ ,  $1.8\pm 0.3$ ,  $1.2\pm 0.6$   $\mu\text{mol/L}$  during the wet season,  
263 respectively, a 3-6 times increase, which were significantly correlated with the flow ( $P$   
264  $< 0.05$ ) (Table. S2), indicating an increase in non-point loading during rainfall events  
265 (Lu et al., 2015). In Lake Erhai, concentrations of DOP, PIP and POP increased from  
266  $0.3\pm 0.2$ ,  $0.3\pm 0.07$  and  $0.5\pm 0.2$   $\mu\text{mol/L}$  during the dry season to  $0.7\pm 0.2$ ,  $2.1\pm 0.4$   
267 and  $2.0\pm 0.5$   $\mu\text{mol/L}$  during the wet season, respectively, a 3-7 times increase. But a  
268 triple decrease in the DIP concentration from  $0.45\pm 0.05$   $\mu\text{mol/L}$  during the dry season  
269 to  $0.11\pm 0.02$   $\mu\text{mol/L}$  during the wet season was observed, which may be attributed to  
270 algal uptake due to a double increase in Chl $a$ . Furthermore, APA during the wet season  
271 was almost double that of the dry season (Fig. S3). SPM was positively correlated with  
272 Chl $a$  and GPP in Lake Erhai and flow in the Miju River ( $P < 0.05$ ) (Table.S2), indicating  
273 increases in SPM during the wet season might be related to plankton biomass in Lake  
274 Erhai and terrigenous input in the Miju River, respectively. Higher WT (average  $21.7$   
275  $\pm 1.7$   $^{\circ}\text{C}$ ) and lower DO ( $4.0\pm 1.4$   $\text{mg/L}$ ) happened during the wet season than that in  
276 the dry season ( $14.4\pm 3.3^{\circ}\text{C}$  for WT;  $7.5\pm 1.1$   $\text{mg/L}$  for DO), while pH changed little  
277 (Fig. S4).

278 The nutrient regimes were very different in the Miju River and Lake Erhai and  
279 gradually changed across the estuary. During the wet season, TP pool shifted from being  
280 dominated by TDP in the Miju River and estuary regions ( $73\pm 5\%$ ) to being dominated  
281 by TPP ( $74\pm 11\%$ ) in Lake Erhai. Especially DIP concentrations decreased dramatically  
282 from the Miju River ( $8.0\pm 0.9\ \mu\text{mol/L}$ ) across the estuary ( $4.1\pm 0.2\ \mu\text{mol/L}$ ), to Lake  
283 Erhai ( $0.14\pm 0.02\ \mu\text{mol/L}$ ), a 60-fold decrease, and TPP pool increased by 1.5-fold from  
284 the Miju River ( $3.0\pm 0.2\ \mu\text{mol/L}$ ) across the estuary ( $2.1\pm 0.4\ \mu\text{mol/L}$ ), to Lake Erhai  
285 ( $4.3\pm 1.6\ \mu\text{mol/L}$ ). Comparison with some other rivers and lakes (Table S3) shows that  
286 the dominant contribution of TPP to TP also occurs in Lakes Huron, Erie, Taihu and  
287 Dianchi (accounting for 54-88%) and in rivers (accounting for 56-91%), such as  
288 Mississippi River, Maumee River, Jourdon River, Chena River, Yellow River and  
289 inflowing rivers of Lake Taihu, etc. Higher TPP abundance indicates a stronger water-  
290 particulate exchange activity (Zhang and Huang, 2011).  $K_d$  values of DIP ( $1.6\pm 0.6$  vs.  
291  $0.03\pm 0.01\ \text{L/mg}$ ) and TDP ( $0.33\pm 0.15$  vs.  $0.05\pm 0.03\ \text{L/mg}$ ) in Lake Erhai are  
292 apparently higher than that in the Miju River ( $P < 0.05$ ) (Fig.1b), indicating the higher  
293 particle-reactivity of SPM in the lake in regulating the abundance, distribution and  
294 cycling of P in lakes, which also happens in other rivers and lakes (Table S3).

### 295 **3.2. P-buffering potential of SPM as a source or sink**

296 The  $K_f$ ,  $\text{EPC}_0$ , the criterion  $\delta$  values and  $K_{d-eq}$  calculated of the isotherms using eq  
297 (3)–(5) for all 11 stations are presented in Fig. 2 and Fig. S5. Results demonstrated that  
298 there were different sorption behaviors of SPM between the Miju River and Lake Erhai  
299 during different sampling months. During the wet seasons, P sorption ability of SPM in

300 the Miju River was higher with larger  $K_f$  values ( $1.1 \pm 0.03$  L/mg) compared to the dry  
301 seasons ( $0.88 \pm 0.07$  L/mg), but the SPM in Lake Erhai showed lower P sorption ability  
302 compared to the dry seasons, with smaller  $K_f$  values ( $0.47 \pm 0.08$  vs.  $0.76 \pm 0.02$  L/mg)  
303 and larger  $EPC_0$  values ( $5.03 \pm 2.69$  vs.  $0.61 \pm 0.65$   $\mu\text{mol/L}$ ). Meanwhile, the SPM in  
304 Lake Erhai showed lower P sorption ability than that in the Miju River, with smaller  $K_f$   
305 values ( $0.47 \pm 0.08$  vs.  $1.1 \pm 0.03$  L/mg) and larger  $EPC_0$  values ( $5.03 \pm 2.69$  vs.  $2.52 \pm 0.69$   
306  $\mu\text{mol/L}$ ).

307 By comparison between DIP and  $EPC_0$ ,  $K_{d-TDP}$  and  $K_{d-eq}$  (TDP) (Zhang and Chi,  
308 2002), respectively, we can predict whether SPM was a source or sink of P in the Miju  
309 River and Lake Erhai (Fig. 2). During the dry season, with relatively low SPM and P  
310 input,  $K_{d-TDP}$  was equal to the  $K_{d-eq}$ , and  $\delta$  values in the river (average  $0.03 \pm 0.02$   $\mu\text{mol/L}$ )  
311 and the lake ( $0.06 \pm 0.03$   $\mu\text{mol/L}$ ) were very low and close to zero, i.e. DIP was roughly  
312 in equilibrium with the SPM. During the wet season, in the Miju river, the  $K_{d-eq}$  (TDP)  
313 was about 40 times higher  $K_{d-TDP}$  (the slope close to zero), and  $\delta$  was positive, and on  
314 average, adsorbed  $2.17 \pm 1.12$   $\mu\text{mol/L}$ , and potentially removed  $45.1 \pm 0.5\%$  of DIP  
315 concentrations, i.e. the SPM played a strong P-sink to buffer additional P loading into  
316 the lake after rainfall. However, for Lake Erhai,  $K_{d-TDP}$  was about triple  $K_{d-eq}$  (TDP), and  
317  $\delta$  was negative, and on average, released P up to  $2.52 \pm 1.14$   $\mu\text{mol/L}$ , accounting for 52.5%  
318 of TPP, causing a 25-fold increase in DIP concentration. This could explain why  
319 “secondary” algal blooms occur under the DIP-limitation in mesotrophic lakes.  
320 Particulates can also contribute to soluble P when the DPS exceeds 25% or 30% (Ohno  
321 et al., 2007). During the wet season, the DPS ( $60.1 \pm 12.1\%$ ) was higher in Lake Erhai

322 as compared to  $16.7 \pm 4.7\%$  in the Miju River, indicating the stronger driving force for  
323 P-release of the SPM in Lake Erhai (Fig. 3). Therefore, the SPM could serve as a strong  
324 P-sink in the river and a P-source in the lake to buffer dissolved P level. Especially, the  
325 risk of SPM as a P-source to lake eutrophication should not be ignored.

### 326 **3.3. Factors influencing P-buffering of SPM**

#### 327 *3.3.1 Effect of particulate organic matter*

328 The P sorption ability of SPM is governed by its chemical composition and  
329 biological activity. Pearson correlation (Table S4) showed P sorption ability of SPM,  
330 characterized by  $\delta$ ,  $K_f$  and  $K_{d-eg}$ , was significantly negatively correlated to POM,  
331 phytoplankton-POC and APA ( $P < 0.01$ ), and positively correlated to C:N and C:P ( $P <$   
332  $0.01$ ), respectively. GAM model further identified (Table 1) that the phytoplankton-  
333 POC, APA and C:N were significant explanatory variables ( $P < 0.001$ ), which could  
334 explain more than 60% variability of  $\delta$  and  $EPC_0$  and more than 30% variability of  $K_f$   
335 and  $K_{d-eg}$ . The mean contributions of phytoplankton-POC, APA and C:N were 79%, 64%  
336 and 49%, respectively. Consequently, the phytoplankton-POC and its biodegradability  
337 (APA, C:N and C:P) were important factors inducing P release from SPM. Previous  
338 studies have confirmed that phytoplankton-POC is strongly related to the number of  
339 carboxylic (-COOH) groups, so P release may be enhanced if monocarboxylic acid  
340 increases (Staunton and Leprince, 1996).

341 This was consistent with the spatiotemporal patterns of POC fractions of SPM in  
342 Lake Erhai and the Miju River (Fig.3, Fig.S3). In Lake Erhai from the dry seasons to  
343 wet seasons, the significantly increased ratio of phytoplankton-POC ( $30 \pm 3\%$  vs.

344 68±11%), the consistently lower organic C:N (23.5±5.5 vs. 8.0±1.6) and C:P  
345 (223.4±56.6 vs. 92.5±20.5) ratios, and the higher APA (0.018±0.006 vs. 0.038±0.006  
346 mmol/L/h) of the SPM can all prove its autochthonous origin from algae and organic  
347 debris of planktonic microorganisms with biolabile compounds for SPM in Lake Erhai  
348 during the wet seasons (called phytoplankton-SPM), indicating greater metabolic  
349 availability and higher turnover of POP by mineralization of microbial activity (Feng  
350 et al., 2018). Therefore, the SPM in the wet season enriched in phytoplankton-POC  
351 helps P release due to the high reactivity in aquatic environments.

352 But for the Miju River, PIM dominated SPM, accounting for 65±14%. The more  
353 abundant detritus-POC (0.55±0.21 mg/L) than phytoplankton-POC (0.24±0.1 mg/L) all  
354 year round and the higher organic C:N (22.6±9.9) and C:P (229±121) ratios of the SPM  
355 indicate its terrestrial plant or soil derived sources (called detritus-SPM) including clay  
356 minerals, silt, insoluble salts and colloidal aggregates (Turner and Millward, 2002),  
357 which may be refractory, inhibiting P release.

### 358 **3.3.2 Effect of P activity of SPM**

359 The P activity of SPM is governed by P species, which reflects P potential  
360 bioavailability transported in particle-water interface (Tang et al., 2018). The P  
361 (NH<sub>4</sub>Cl+BD) fraction is fully mobile under natural conditions (Yi et al., 2017), and the  
362 POP pool, such as NaOH<sub>25</sub>-nrP, also release P from polyphosphate-incorporating  
363 bacteria mineralization or bacterial respiration (Hupfer et al., 2004; Zhang et al., 2020).  
364 Consequently, the fractions of NH<sub>4</sub>Cl-P, BD-P, and NaOH<sub>25</sub>-nrP were all considered as  
365 the labile or bioavailable fractions (BAP) in this study.

366 Significant correlation occurred between P fractions and adsorption parameters  
367 across the sampling sites (Table S4). BAP fractions were significantly negatively  
368 correlated with P sorption ability of SPM, characterized by  $\delta$ ,  $K_f$  and  $K_{d-eq}$ , and  
369 positively correlated with  $EPC_0$ ; while inert fractions (HCl-P and NaOH<sub>25</sub>-rP) of SPM  
370 were positively correlated with  $\delta$ ,  $K_f$  and  $K_{d-eq}$ , and negatively correlated with  $EPC_0$ ,  
371 respectively. GAM model (Table 1) further confirmed that NaOH<sub>25</sub>-nrP, NH<sub>4</sub>Cl-P,  
372 NaOH<sub>25</sub>-rP and HCl-P were identified as significant explanatory variables ( $P < 0.001$ ),  
373 which could explain 46.8%~90.6%, 50.1%~83.7%, 33.1%~82.8% and 31.8%~79.2%  
374 of variations in the P sorption ability of SPM, and explain 84.5%, 90.2%, 73.6% and  
375 55.3% variations of  $EPC_0$ , respectively. Therefore, the P release capacity of SPM could  
376 be attributed to the proportions of BAP; while for P adsorption capacity of SPM, the  
377 aluminum and calcium oxides might be a key driver rather than iron compounds (Kerr  
378 et al., 2011). As shown in Fig.2 and Fig.3, in Lake Erhai, the BAP concentrations  
379 increased from  $479 \pm 105$  mg/kg to  $1260 \pm 210$  mg/kg leading to higher P-release  
380 potential from  $0.17 \pm 0.07$   $\mu\text{mol/L}$  to  $5.72 \pm 1.84$   $\mu\text{mol/L}$ ; while in the Miju River, P-  
381 adsorption potential increased from  $0.07 \pm 0.06$   $\mu\text{mol/L}$  to  $5.97 \pm 1.12$   $\mu\text{mol/L}$  as the inert  
382 fractions increased from  $205 \pm 30$  mg/kg to  $539 \pm 112$  mg/kg. Consistently, in other  
383 well-known rivers (Rydin, 2000), the higher proportion of inert fractions in SPM, such  
384 as Nile and Mississippi Rivers (54%~57%), the Amazon River (67%~72%), and the  
385 Yellow River (82%), could reflect its higher P adsorption potential (Pan et al., 2013;  
386 Sutula et al., 2004).

387 P adsorption experiment (Fig.4) also confirmed that P adsorption amount ( $Q_e$ ) of

388 the phytoplankton-SPM in Lake Erhai (-11.5~13.3 mg/kg) was much lower than that of  
389 the detritus-SPM in the Miju River (-1.8~70.9 mg/kg). In Lake Erhai, this quasi-  
390 positive correlation among BAP fractions, phytoplankton-POC and  $Q_e$  revealed that the  
391 low  $Q_e$  might be attributed to BAP fractions and phytoplankton-POC, as described by  
392 [Zhu et al. 2015](#) that phytoplankton-POC complexes with surface-bound Fe to form  
393 soluble organic-metal compounds can replace and release previously-sorbed P by  
394 competitive adsorption or ligand exchange. Furthermore, this quasi-negative  
395 correlation between APA and  $Q_e$  suggests the higher APA could stimulate visible P-  
396 release of phytoplankton-SPM. However, in the Miju River, the inert fractions  
397 significantly increased from 27% to 54% with increasing  $Q_e$  of detritus-SPM, indicating  
398 P could be resorbed and immobilized to enriched Al/Ca metal oxides. Similarly, for the  
399 Yellow River, the elevated concentrations of Al(OH)<sub>3</sub> and Ca metal oxides in particles  
400 can prevent P release by adsorbing P due to their stability under both oxic and anoxic  
401 conditions ([Huser et al., 2016](#)).

402 In addition, the DO and WT were identified as significant explanatory variables  
403 ( $P < 0.001$ ), explaining 40~67.8% of the variations in the  $K_{d(DIP)}$ ,  $K_{d(DOP)}$  and  $K_{d(TDP)}$   
404 ([Table. 1](#)).  $K_{d(DIP)}$  and  $K_{d(DOP)}$  of SPM were significantly positively correlated with DO,  
405 and negatively correlated with WT ([Table. S4](#)), indicating that higher WT ( $21.7 \pm 1.7$  °C)  
406 and anoxia (DO<5 mg/L) during the wet season can stimulate P release of SPM.  
407 Consequently, P exchange in the particle-water interface is largely controlled by POC  
408 and P species of SPM and water environment. Nevertheless, under which conditions  
409 the coupling of POC and metal oxyhydroxides or metal ions affects the P release and

410 how fast the release happens needs to be further studied.

### 411 **3.4. Effect of increasing SPM on P-buffering**

412 Results from the adsorption isotherm experiment by changing mixing ratio of  
413 phytoplankton-SPM and detritus-SPM showed that P sorption capacity dropped down  
414 as the phytoplankton-SPM proportion increased, with smaller  $Q_e$  and  $K_f$  and higher  
415  $EPC_0$  (Fig.5a). The phytoplankton-SPM showed the minimum  $Q_e$  (-11.5~13.2 mg/kg)  
416 and  $K_f$ (0.35 L/mg) and maximum  $EPC_0$  (5.11 mg/L); while the detritus-SPM showed  
417 the maximum  $Q_e$  (20.1~70.9 mg/k) and  $K_f$ (0.95 L/mg) and the minimum  $EPC_0$  (0.15  
418 mg/L). These results confirmed significant differences in P sorption capacity between  
419 the phytoplankton-SPM and detritus-SPM ( $P<0.05$ ).

420 Moreover, when detritus-SPM concentrations increased from 2 to 35 mg/L, DIP  
421 removal proportion increased up to more than 50% in the Miju River, but decreased to  
422 below -50% (indicating P-release) when phytoplankton-SPM concentrations increased  
423 from 2 to 35 mg/L in Lake Erhai (Fig. 5 b, c). Especially when  $C_0 \leq 5 \mu\text{mol/L}$  and  
424 phytoplankton-SPM  $\geq 16 \text{ mg/L}$ , the release amount of P could lead to 0.5- and 3-fold  
425 increase in the DIP flux; While when detritus-SPM  $\geq 16 \text{ mg/L}$ , P removal exceeds 30%  
426 at  $C_0 \geq 5 \mu\text{mol/L}$  and 50% at  $C_0 \geq 7 \mu\text{mol/L}$ . Consequently, the higher phytoplankton-  
427 SPM concentration could release more DIP, especially at lower DIP concentration,  
428 while the higher detritus-SPM concentration could remove more DIP, especially at  
429 higher DIP concentration. The conclusion again reflected P-buffering of  
430 phytoplankton-SPM as a P-source and detritus-SPM as a P-sink, which may also apply  
431 to other rivers and lakes rich in SPM (Table S3), such as Lake Dianchi with the  $29.6 \pm 8$

432 mg/L SPM concentration, and the Yellow River, the inflowing rivers to Lakes Taihu  
433 and Dianchi with a range of SPM concentration from 27 to 656.3 mg/L.

434 The relationships between  $K_{d-eq}$  (DIP) and SPM concentration were fitted at  
435 different initial DIP levels in Lake Erhai and the Miju River (Fig.5 d, e). In Lake Erhai,  
436 the significantly inverse correlation between  $K_{d-eq}$  (DIP) and phytoplankton-SPM ( $P <$   
437  $0.01$ ) demonstrated a strong “particle concentration effect” (Lin and Guo, 2016),  
438 indicating higher phytoplankton-SPM concentration enhances P-buffering. However,  
439 in the Miju River, the inverse correlation between  $K_{d-eq}$  (DIP) and detritus-SPM is not  
440 significant, demonstrating the P adsorption capacity of detritus-SPM was relatively  
441 stable (Lin et al., 2012).

442 We used the modified  $K_d$  model with realistic estimates of the input parameters,  
443 and biological uptake of P from the relationship between TP and GPP reported by Smith  
444 (1979) to determine the relationship of SPM and TDP (Text S4, Fig.6). We hypothesise  
445 that the removal of dissolved P from aquatic ecosystems mainly reflects the  
446 combination of biological uptake and particle adsorption (Huang and Zhang, 2010).

447 For similar initial and final values of GPP (based on simulations over the range  
448  $A_2 - A_1 = 0 - 0.9 \text{ g/m}^3/\text{d}$ ) i.e., moderate or negligible biological uptake of P in the Miju  
449 River, the model predicted that TDP removal would increase significantly as SPM  
450 concentration increased, mainly because of adsorption by physicochemical control  
451 (Fig.6a). The SPM could therefore shift from a source to sink of P, depending on its  
452 concentrations and  $K_d$ . In the Miju River, during the wet season, the detritus-SPM  
453 concentration averaged 10.3 mg/L and was above the critical thresholds of 7.2 mg/L at

454 average  $K_d$  values of 0.07, suggesting the detritus-SPM was a P-sink, which was  
455 consistent with the adsorption isotherms. If TDP inputs continued to increase (i.e., the  
456  $K_d$  decline), more SPM would be needed to buffer the additional TDP loading.

457 However, if GPP is much higher at the end than at the beginning ( $A_2 - A_1 > 0.9$   
458  $\text{g/m}^3/\text{d}$ ) i.e., biological uptake of P dominates, as algae biomass sharply increase in Lake  
459 Erhai during the wet season. Interestingly, the TDP removal decreased sharply as the  
460 SPM concentration increased (Fig.6b). i.e., autotrophs can facilitate the accumulation  
461 of P on phytoplankton-SPM and enhance P lability, which in turn increases the  
462 bioavailable P for reuse, thereby promoting outbreak of algal blooms (Xie et al., 2003).

463 Both experiments and modeling confirmed that increasing SPM concentrations  
464 could enhance P-buffering. Especially the phytoplankton-SPM to buffer DIP-limitation  
465 during algae blooms should not be ignored, not just focused on eliminating sediments  
466 loading through sediment capping, sediment dredging (Liu et al., 2016; Wen et al.,  
467 2020).

### 468 **3.5. Implications of SPM for P cycling during transport from river to lake**

469 We drew a mechanism diagram of P-buffering of SPM during transport in the  
470 river-lake system (Fig.7). From the dry season to wet season, the increased SPM along  
471 with more P nutrient from terrestrial sources flows to the rivers and lakes due to the  
472 rainfall and runoff increase (Zhang et al., 2014). For rivers, the detritus-SPM enriched  
473 with PIM adsorbs P to form inert P fractions ( $K_d$  increase), which tend to sedimentation  
474 rather than re-release into water, thus could reduce 45% of P input to lakes. Global  
475 warming will increase the chemical and physical weathering rates, and the rainfall will

476 increase weathered particles content and agricultural nonpoint sources into rivers (Pan  
477 et al., 2013). However, the detritus-SPM content was not enough to fully buffer the  
478 increased loading. Therefore, plenty of P loading into lakes at the proper temperature  
479 ( $>20^{\circ}\text{C}$ ) can trigger phytoplankton growth, leading to the increase of phytoplankton-  
480 SPM from algae metabolism. When the DIP is insufficient to satisfy the algae growth,  
481 the increasing phytoplankton-SPM enriched with BAP fractions could release more  
482 DIP ( $K_d$  decrease), causing a 25-fold increase for algae reuse, which creates a vicious  
483 cycling among algae-SPM-P to worsen water quality. Furthermore, anoxic, higher WT  
484 and enhanced APA in the wet season had positive effects on P liberation from the  
485 phytoplankton-SPM. This could explain why algal blooms in Lake Erhai can persist  
486 until November under DIP-limitation. The SPM shifted from a P-sink to a P-source to  
487 buffer P levels and promote P cycling during transport from rivers to lakes.

#### 488 **4. Conclusions**

489 Our study offers a rare insight into two distinctive P cycling regimes between  
490 rivers and lakes, and provides new evidences of SPM as a P-sink in rivers and a P-  
491 source in lakes to buffer the level and partition of P, which was closely linked to the  
492 quality and quantity of SPM. The seasonal differences in POM components and P  
493 activity of SPM can influence P balance in the SPM-water interface, i.e., from the dry  
494 seasons to wet seasons, the significantly increased phytoplankton-POC and BAP  
495 fractions of SPM induced more P-release from SPM. Moreover, increasing SPM  
496 concentrations could enhance the P-buffering effect. When phytoplankton-SPM  $\geq 16$

497 mg/L at  $C_0 \leq 5 \mu\text{mol/L}$ , P release increased by 50%-300%; when detritus-SPM  $\geq 16$   
498 mg/L at  $C_0 \geq 5 \mu\text{mol/L}$ , P removal could exceed 30%. This study highlights the  
499 importance of SPM as a sink or source of P in regulating P cycling cannot be ignored,  
500 and helps water managers to rethink the inner loop among algae-SPM-P under DIP-  
501 limitation.

## 502 **Supplementary information**

503 Supplementary information contains Figures S1–S7, Tables S1–S4, and six parts:  
504 Text S1–Text S6.

## 505 **Acknowledgments**

506 This research was supported by National Natural Science Foundation of China  
507 (No. U1902207), Open fund project of Yunnan Key Laboratory of Pollution Process  
508 and Management of Plateau Lake-Watershed (No. 2020-02-2-W2, 2020-02-2-W3).

## 509 **Declaration of competing interest**

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

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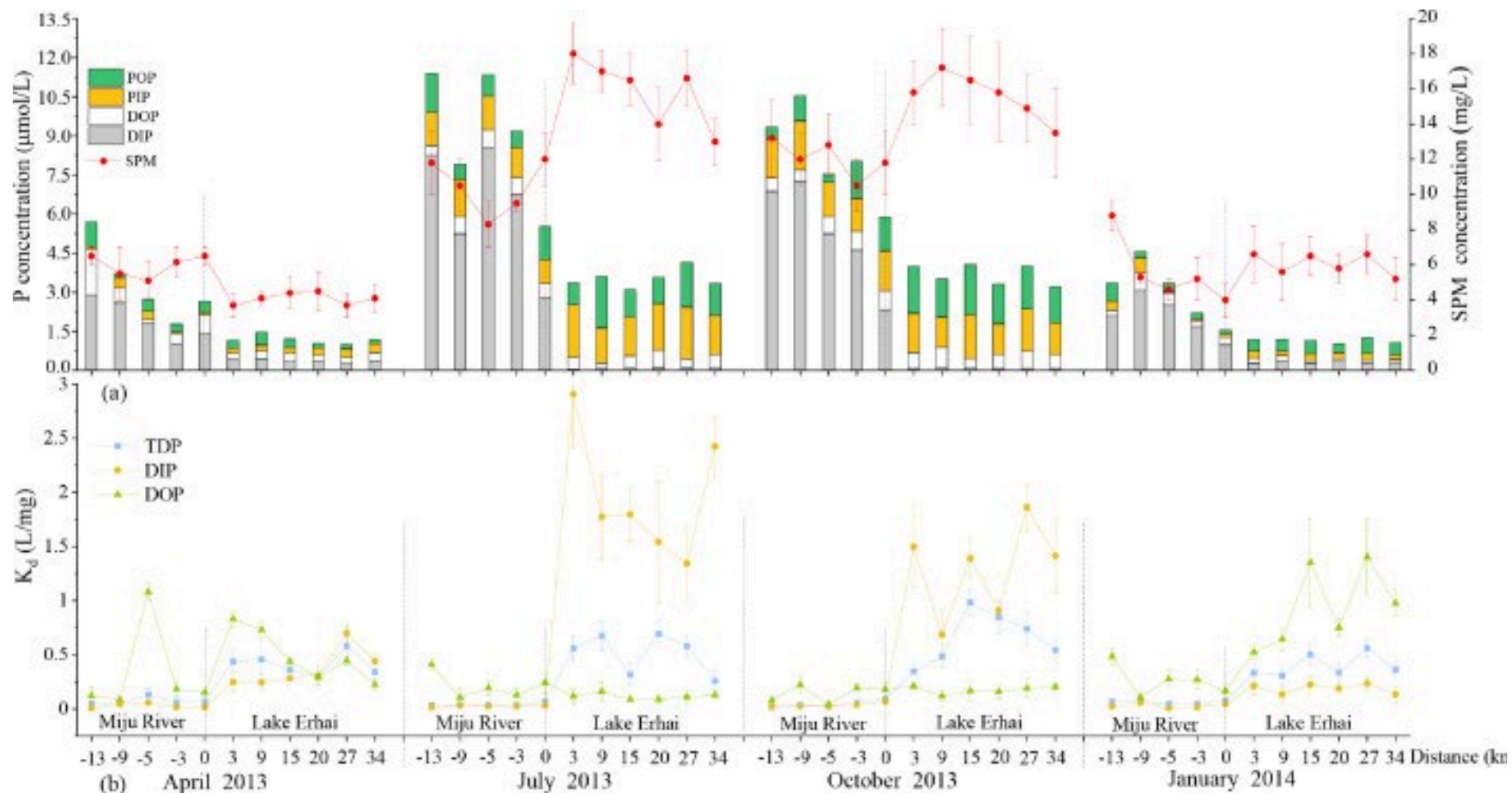
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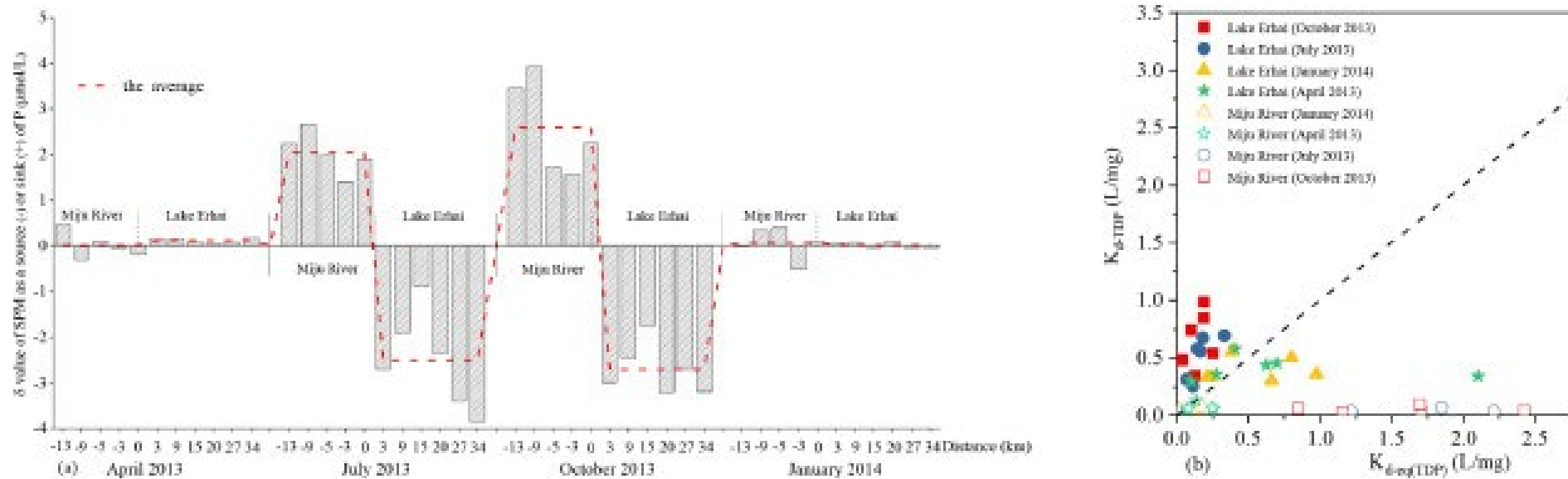
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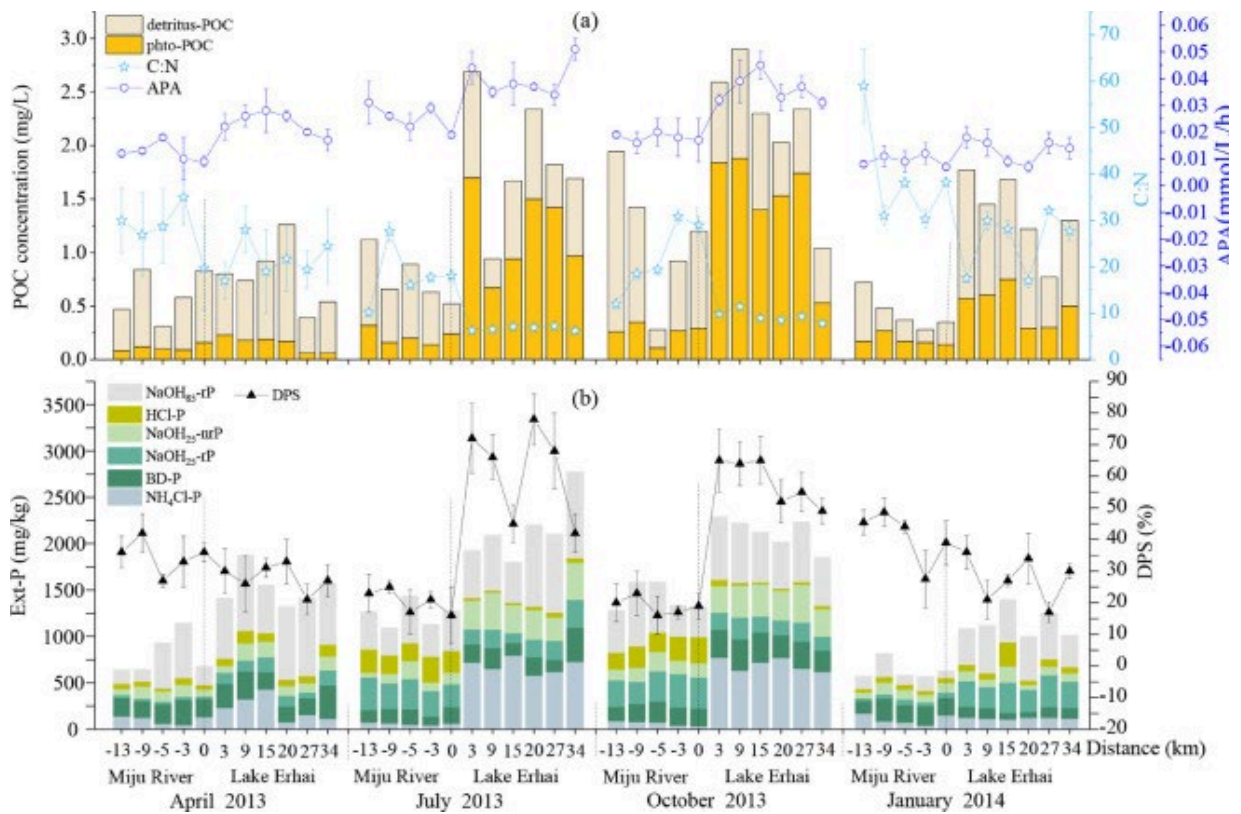
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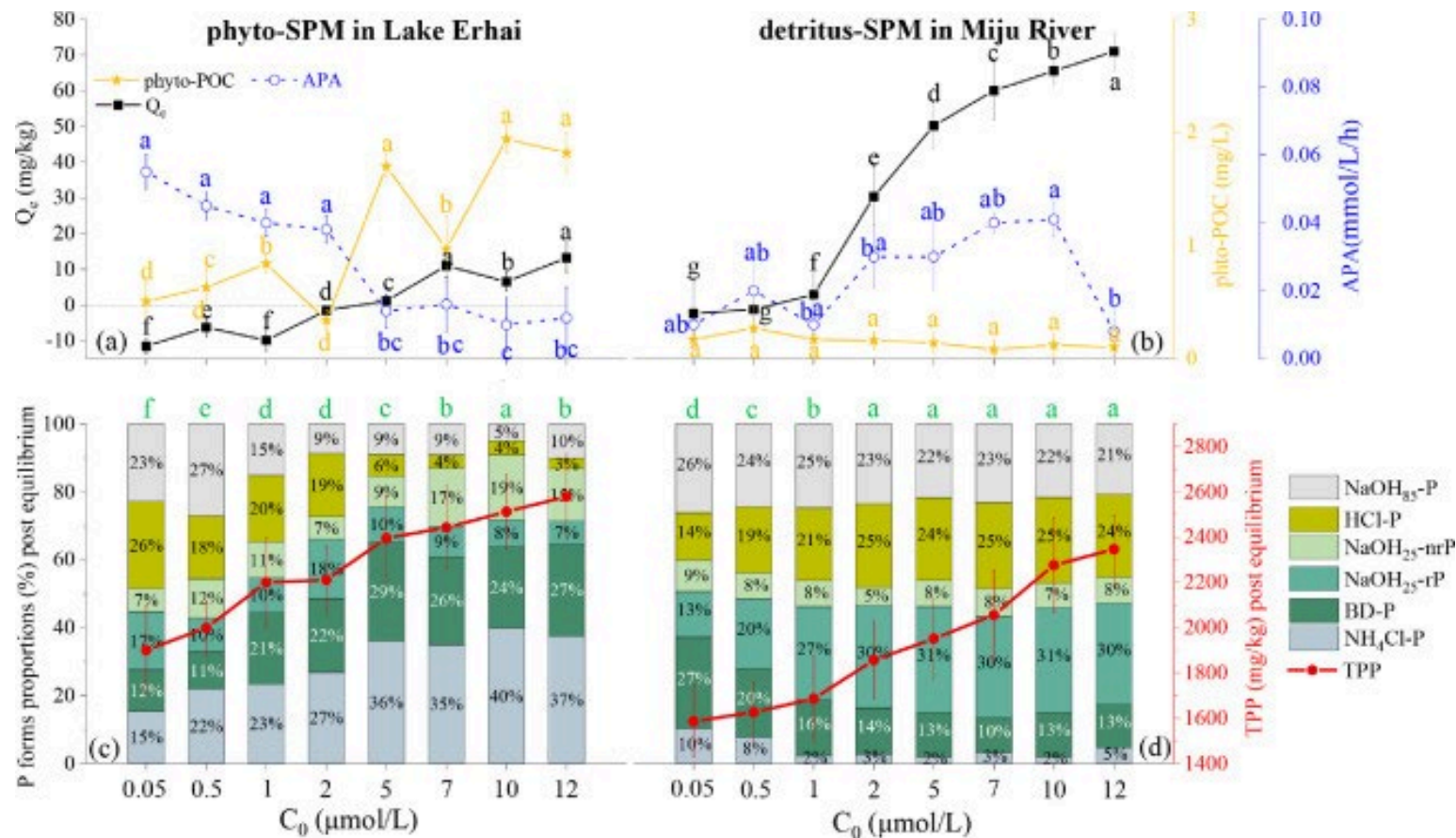
**Fig. 1.** (a) Spatial-temporal distribution of P species and suspended particulate matter (SPM) and (b) average partition of different P forms across the transition from the Miju River to Lake Erhai during different sampling months. The dashed line represents the estuary at the river-lake interface.



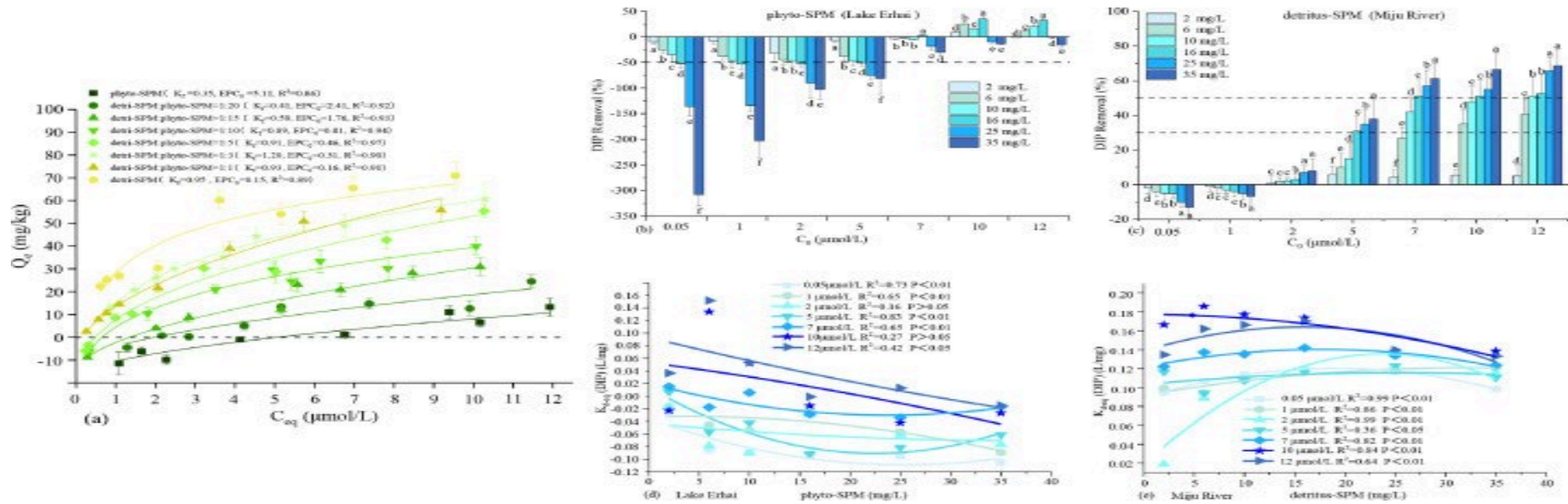
**Fig.2.** (a) The calculated potential of each SPM as a sink or source of P calculated by  $\delta$  in the Miju River and Lake Erhai during different sampling months. (b) Correlations between  $K_{d-eq}$  (TDP) and  $K_{d-TDP}$  in the Miju River and Lake Erhai during different sampling months.



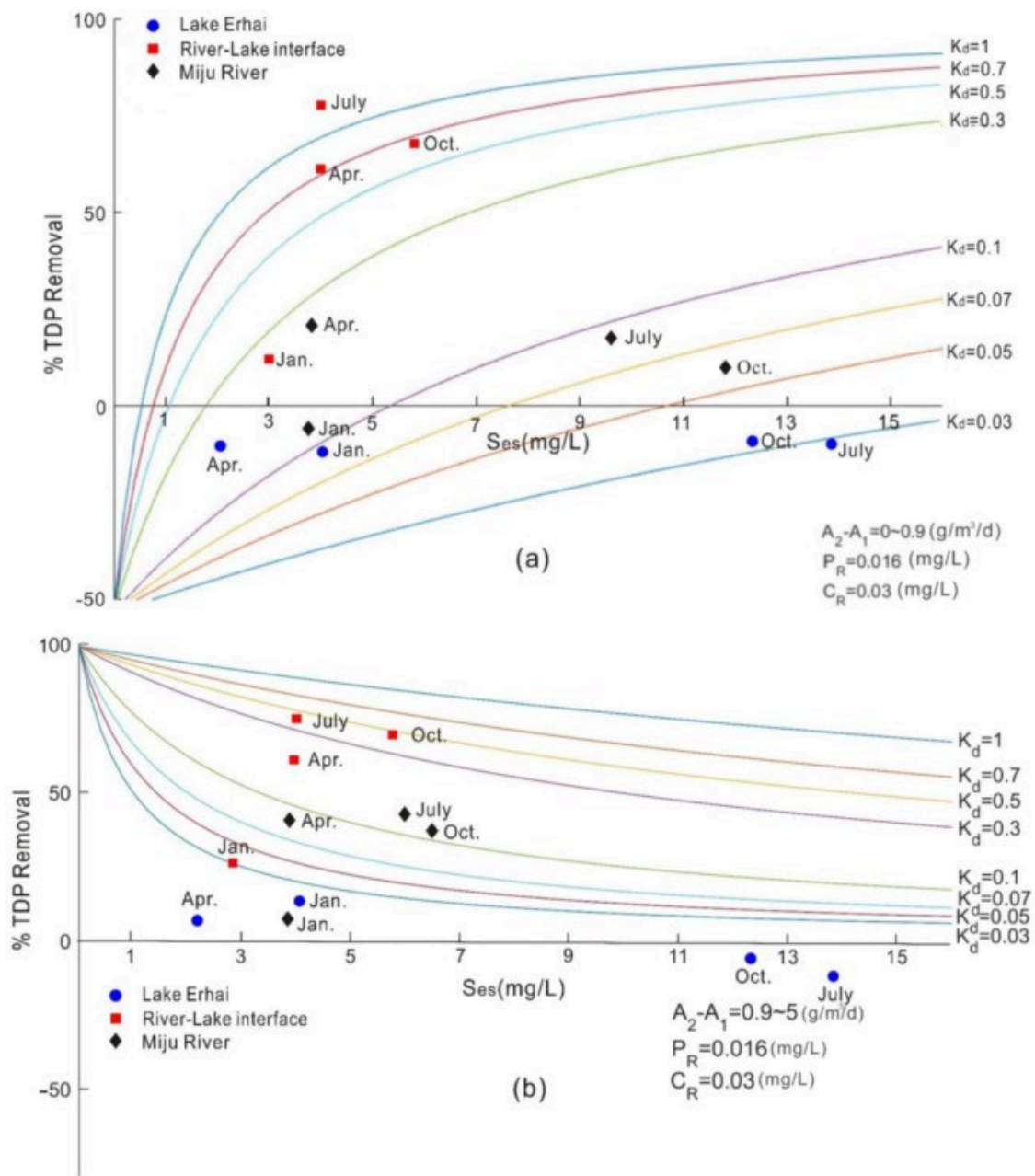
**Fig.3.** (a) Spatial-temporal variations in the POC from phytoplankton (phyto-POC) and detritus (detritus-POC), the organic C:N ratio in SPM and alkaline phosphatase activity (APA); (b) Different P species in SPM and the degree of phosphorus saturation (DPS) from the Miju River to Lake Erhai during different sampling months.



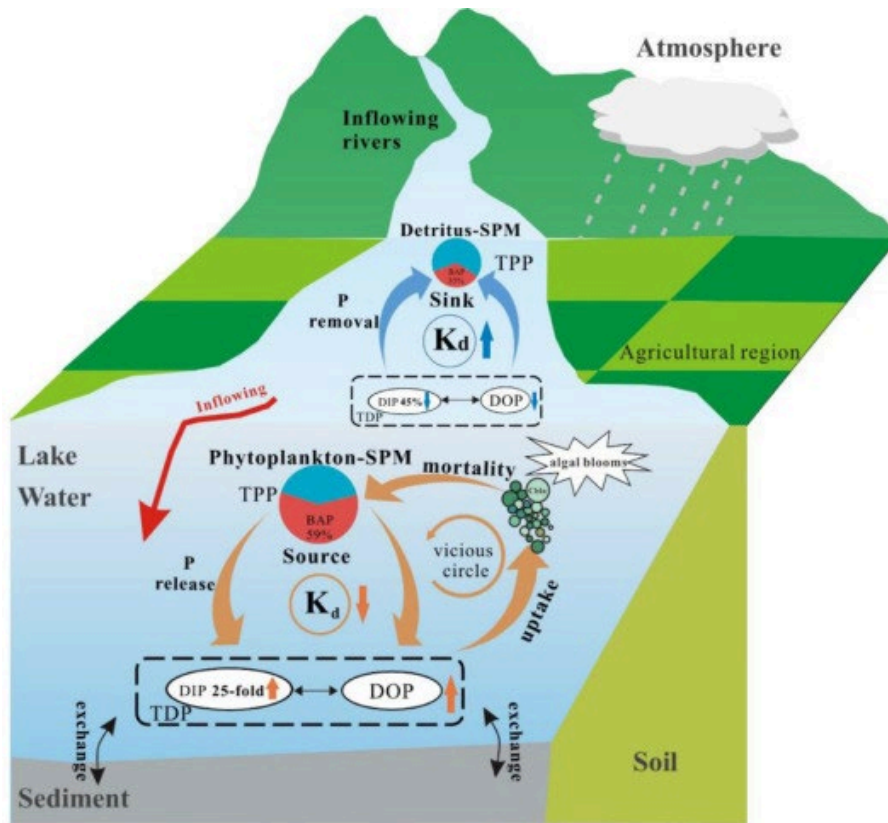
**Fig.4.** Variations of the P sorption capacity ( $Q_e$ ), phyto-POC, APA and proportions of particulate P species with at equilibrium at different initial DIP concentrations ( $C_0$ ) in Lake Erhai (a, c) and the Miju River (b, d) by adsorption isotherms of SPM. The phyto-SPM came from the station LE-2 in Lake Erhai, and the detritus-SPM came from the station MR-1 in the Miju River in October. The different letters (i.e., a–c) indicate significant differences among  $C_0$  (one-way ANOVA,  $p \leq 0.05$ ). Different letters in green represent significant differences of BAP fractions ( $\text{NH}_4\text{Cl-P} + \text{BD-P} + \text{NaOH}_{25}\text{-nrP}$ ) (c) and inert fractions of  $\text{HCl-P} + \text{NaOH}_{25}\text{-rP}$  (d) among  $C_0$ .



**Fig.5.** (a) P adsorption isotherms by the mixture of detritus-SPM and phyto-SPM, and changed the mixture mass ratio from 1:1 to 1:20; The detritus-SPM came from the station MR-1 in the Miju River in October, while phyto-SPM was obtained by adding fresh algae. The influence of different SPM concentrations (2, 6, 10, 16, 25 and 35 mg/L) on DIP removal (%) and  $K_{d-eq}$  (DIP) at different initial DIP concentration in Lake Erhai (b, d) and the Miju River (c, e). The different letters (i.e., a–c) above the bars indicate significant differences among SPM concentrations (one-way ANOVA,  $p \leq 0.05$ ).



**Fig.6.** Model predicted % removal (+) or release (-) of TDP as a function of the SPM concentrations ( $S_{es}$ ) at various values of  $K_d$  at low algal biomass (a) and high algal biomass (b).  $K_d$  and  $S_{es}$  values in the simulations are consistent with realistic changes in the Miju River and Lake Erhai. The values of the other parameters were the mean of realistic estimates.



**Fig.7.** Mechanism diagram of P buffering effect of SPM as a P-sink in rivers and P-source in lakes on P cycling and  $K_d$  during transport in the river-lake system.

**Table 1.** The GAM fitting results between major factors and parameters ( $\delta$ ,  $EPC_0$ ,  $K_f$  and  $K_{d-eq}$ ) of Freundlich crossover-type equations calculated by a nonlinear fit for the P adsorption isotherms of SPM; and between major factors and  $K_d$  ( $K_{d(DOP)}$ ,  $K_{d(TDP)}$  and  $K_{d(DIP)}$ ) in field investigation in the Miju River and Lake Erhai.

Response variables	Explanatory variable	Edf	$F$	$p$	Dev.expl (%)	$R^2$
$\delta$	NaOH <sub>25</sub> -nrP	3.59	85.98	$<2 \times 10^{-16}$ ***	90.6%	0.898
	NH <sub>4</sub> Cl-P	4.14	39.04	$<2 \times 10^{-16}$ ***	83.7%	0.824
	NaOH <sub>25</sub> -rP	4.07	37.14	$<2 \times 10^{-16}$ ***	82.8%	0.813
	APA	6.77	20.84	$<2 \times 10^{-16}$ ***	82.3%	0.795
	Phyto-POC	5.68	22.39	$<2 \times 10^{-16}$ ***	80.9%	0.779
	C:N	3.52	18.41	$4.17 \times 10^{-10}$ ***	67.2%	0.643
	HCl-P	3.01	8.45	$5.57 \times 10^{-5}$ ***	46.2%	0.422
	NaOH <sub>85</sub> -P	5.84	3.07	0.0115 *	40.4%	0.306
	POM	2.01	9.25	0.000262 ***	36.8%	0.337
	C:P	2.05	8.48	0.000459 ***	35.4%	0.321
$EPC_0$	NH <sub>4</sub> Cl-P	8.45	35.19	$<2 \times 10^{-16}$ ***	90.2%	0.879
	NaOH <sub>25</sub> -nrP	7.39	22.84	$<2 \times 10^{-16}$ ***	84.5%	0.813
	APA	5.59	24.46	$<2 \times 10^{-16}$ ***	82.8%	0.793
	Phyto-POC	4.29	23.94	$7.8 \times 10^{-15}$ ***	77.1%	0.746
	NaOH <sub>25</sub> -rP	4.47	19.09	$3.72 \times 10^{-12}$ ***	73.6%	0.705
	C:N	3.61	16.41	$2.9 \times 10^{-9}$ ***	65.4%	0.622
	HCl-P	3.08	12.18	$1.03 \times 10^{-6}$ ***	55.3%	0.519
	C:P	5.46	5.07	0.00039***	49.4%	0.42
$K_f$	POM	2.15	9.44	0.000149***	39.5%	0.364
	HCl-P	0.22	17.34	$5.66 \times 10^{-14}$ ***	79.2%	0.752
	Phyto-POC	6.39	3.27	0.00911 **	62.6%	0.325
	NaOH <sub>25</sub> -rP	3.61	4.02	0.00778 **	53.1%	0.274
	NH <sub>4</sub> Cl-P	1.35	9.53	0.000651 ***	50.1%	0.278
	APA	3.21	3.64	0.0139 *	49.4%	0.237
	NaOH <sub>25</sub> -nrP	2.79	3.67	0.0145 *	46.8%	0.218
	C:N	6.63	1.16	0.324	36.5%	0.131
C:P	3.34	2.07	0.102	21.3%	0.144	
$K_{d-eq}$	Phyto-POC	6.11	379.70	$<2 \times 10^{-16}$ ***	97.0%	0.951
	NH <sub>4</sub> Cl-P	3.21	31.93	$<2 \times 10^{-16}$ ***	64.9%	0.604
	BD-P	6.36	4.98	$2.98 \times 10^{-6}$ ***	56.0%	0.564
	NaOH <sub>25</sub> -nrP	5.97	12.25	$1.47 \times 10^{-11}$ ***	54.8%	0.515
	APA	7.51	3.79	0.000658 ***	41.1%	0.373
	NaOH <sub>25</sub> -rP	7.50	3.89	0.000658 ***	33.1%	0.283
	HCl-P	5.59	5.48	$6.61 \times 10^{-5}$ ***	31.8%	0.263
	C:N	4.58	5.14	$7.44 \times 10^{-5}$ ***	30.1%	0.233
	PIM	3.79	5.08	$6.61 \times 10^{-5}$ ***	28.4%	0.219

Field Investigation	K <sub>d</sub> (TDP)	DIP	4.851	22.03	1.25e-14 ***	77.40%	0.746
		DO	6.629	8.571	4.76e-07 ***	66.50%	0.604
		Chla	6.299	7.909	2.2e-06 ***	62.70%	0.563
		SPM	6.17	7.298	6.83e-06 ***	60.70%	0.541
		NH <sub>4</sub> Cl-P	2.138	20.41	3.05e-08 ***	58%	0.558
		HCl-P	6.09	6.024	6.06e-05 ***	56.40%	0.492
		NaOH <sub>25</sub> -rP	4.425	8.08	1.05e-05 ***	55.30%	0.501
		POM	3.61	9.779	4.4e-06 ***	53.80%	0.496
		Phyto-POC	3.762	8.851	1.2e-05 ***	52.20%	0.477
		NaOH <sub>25</sub> -nrP	2.681	10.1	2e-05 ***	46.60%	0.431
		WT	3.693	6.126	0.000401 ***	43.80%	0.385
		TPP	4.469	4.685	0.00132 **	42.50%	0.359
		APA	3.658	6.078	0.000445 ***	42.50%	0.372
	K <sub>d</sub> (DIP)	NaOH <sub>25</sub> -nrP	6.652	26.26	<2e-16 ***	85%	0.823
		C:N	5.552	30.98	<2e-16 ***	84.70%	0.824
		NH <sub>4</sub> Cl-P	4.92	32.87	<2e-16 ***	83.90%	0.819
		DIP	8.224	17.68	1e-15 ***	82.30%	0.781
		Chl a	7.475	19.2	<2e-16 ***	82.20%	0.784
		Phyto-POC	6.656	19.09	1.38e-15 ***	80.70%	0.772
		APA	3.971	29.78	<2e-16 ***	79.20%	0.771
SPM		7.049	12.46	1.79e-10 ***	74.30%	0.693	
DO		6.872	9.006	1.68e-07 ***	67.80%	0.616	
NaOH <sub>25</sub> -rP		6.236	4.338	0.00113 **	49%	0.403	
K <sub>d</sub> (DOP)	TPP	2.675	10.87	9.21e-06 ***	48.80%	0.455	
	WT	6.326	3.764	0.00339 **	45.20%	0.358	
	PIP	1.798	12.1	4.21e-05 ***	40.50%	0.379	
	DOP	5.526	25.35	<2e-16 ***	81.8%	0.791	
	WT	6.797	7.29	4.14e-06 ***	62.60%	0.556	
	PIP	6.945	5.132	0.000186 ***	55.40%	0.469	
	TPP	6.266	4.687	0.00054 ***	51.50%	0.433	
	DO	2.023	10.18	9.44e-05 ***	40%	0.37	
	BD-P	2.779	4.451	0.0066 **	30.10%	0.252	

\*  $p \leq 0.05$ , \*\*  $p \leq 0.01$ , \*\*\*  $p \leq 0.001$