Different forms of phosphorous transformation and release prediction with environment factor in sediments from Lake Dongting, China

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ABSTRACT

Lake Dongting is the largest freshwater lake of China. Because of eutrophication, it has become a hot topic of water environmental protection in China. Although the input of exogenous phosphorus (P) is controlled, the release of endogenous P can maintain eutrophication for an extended time. This study explored the relationship between different environmental factors and the release of endogenous P in sediments through a large-scale field investigation. The migration of endogenous P from sediment to the overlying water layer follows a three-phase cycle process, in which iron and aluminum phosphorus (Fe/Al–P), calcium phosphorus (Ca–P), and TP transformation will occur. Fe/Al–P is the most important source of phosphorus release. The content of total P in Lake Dongting shows the following gradient East Dongting Lake > West Dongting Lake > South Dongting Lake. The contents of Fe/Al–P in East Dongting Lake were highest, which may represent a risk of P release. Therefore, canonical variate analysis showed that the effect relationship of phosphorus release followed ORP > pH > T > depth > conductivity ≈ TDS. Among these, ORP was the most important driving factor, as it affects the conversion of Fe^{3+}–P and Fe^{2+}–P. Finally, a model to predict P release was developed and the P release rate (Ri) was Ri = 0.472 + 0.085[T] + 0.029[TDS] + 0.058 [Cond.] − 0.207[ORP] − 0.134[DO] − 0.047[Depth]. These results have important guiding significance for the selection and implementation of P control technologies.

Keywords: Sediment; Phosphorus transformation; Phosphorus release; Release prediction; Environmental factor; Lake Dongting

1. Introduction

Sediments are important constituents of lake environments as they contain nutrients and are particularly important for geochemical processes. Specifically, surface or near-surface reactions control both the migration and transformation of relevant nutrients [1]. Changes in environmental factors, such as resuspension of nutrients, pH, DO, T, and ORP, directly affect the release of phosphorus (P) [2]. Phosphorus is often the predominant determinant of the degree of lake eutrophication. Even when P input is controlled, internal recycling of endogenous P can maintain eutrophic conditions for extended periods [3–6]. The eutrophication of Tai Lake (in China) has been sustained for several decades since the implementation of a P input ban in 1998. In Loosdrecht (Netherlands), eutrophication continued to occur since 1984, when the input of P was decreased to historically low amounts [7,8]. Endogenous P remains the critical factor for eutrophication; however, large-scale on-site research that investigated the effects

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of environmental factors on the release of endogenous P from sediments is limited [9].

Investigations of endogenous P releases from sediments began as early as the 1940s. Prominent foci were the sources of P deposited in sediments [10], the distribution of P particulates and pore waters in sediments [11], the sequential extraction of different forms of P from sediments [12], and laboratory studies on effects of environmental factors on P in sediments [13]. Currently, the methods to study various forms of P in lake sediments have been well developed. In 1996, the European Standard Test and Measurement Organization launched a joint project for the development of a standard method for the extraction of various forms of P. This organization compared the results of four widely accepted sequential extraction methods (i.e., the Williams method, the Hieltjes–Lijklema method, the Golterman method, and the Ruttenberg method). Eventually, for a continuous method the extraction of different forms of freshwater sediment P was developed: the SMT method [14].

Based on this method, indoor environmental factor simulation experiments were conducted to investigate both migration and transformation processes of different forms of P in sediments. Releases of endogenous P were found to be controlled by a series of physical, chemical, and biological processes [15]. The factors that affect P releases include disturbance, oxidation–reduction potential (ORP), and pH. Disturbances of equilibria and gradients during the resuspension of sediments could result in the absorption of endogenous P by particles [16]. Therefore, for an accurate understanding of all processes involving P that occur in lake sediments, the mechanisms underlying endogenous P release need to be further studied. In particular, the results of laboratory studies need to be combined with on-site measurements. While observations that can be connected with on-site measurements are more complicated, they also provide more accurate information on uncontrolled and difficult to study processes compared with individual processes. In contrast, the results of laboratory studies often cannot represent the natural work, since individual processes are often controlled and can thus be accurately described by using experimental manipulations. The present study reconciles the results of both types of studies and generates combined results that maximize the benefits of each type of approach while minimizing their limitations.

The objectives of this study were to understand the changes among various forms of P. These include dissolved inorganic phosphorus (DOP), dissolved total phosphorus (DTP), organic phosphorus (OP), inorganic phosphorus (IP), iron and aluminum phosphorus (Fe/Al–P), and calcium phosphorus (Ca–P). The characteristics of release and distribution of the various forms were described for sediments, pore water, and overlying water. These objectives were accomplished by using large-scale on-site measurements, which were merged with the results of more controlled studies. This approach facilitates accurate descriptions of individual processes that occur in the sediments of lakes. Furthermore, the results were used to develop a model based on environmental factors that could be easily and quickly measured such as hydrodynamics and physical characteristics of sediments. Using this predictive model, it became possible to predict the released amounts of P from sediment in a timely manner, which helps to develop strategies for the remediation of eutrophication. Once validated, this model can be used to rapidly test different strategies for the control of P releases. This may lead to the complete restoration of lakes from detrimental effects of cultural eutrophication.

2. Materials and methods

2.1. On-site measurements

Lake Dongting (or Grotto Court Lake) is a typical water-throughput lake located in northern Hunan Province, China (29°200″N, 112°550″E). It is the largest freshwater lake of China, with a surface area that ranges from 2,740 km² during dry periods to more than 12,000 km² during wet periods. The mean depth of the lake is 6.7 m, while the maximum depth is 30.8 m. Based on the rates of sediment accumulation and the history of land reclamation, Lake Dongting has been conceptually divided into three regions: East, South, and West Dongting [17].

In April 2019 and September 2019, a Horiba 50-U meter (Horiba, Kyoto, Japan) was used to measure temperature (T), dissolved oxygen (DO), pH, total dissolved solids (TDS), ORP, and water depth (WD) about 5 cm above the sediment surface. Representative transects with 5–20 stations were defined in East, South, and West Dongting. A total of 43 sampling points were established and four years of meteorological data (from the Hunan Meteorological Institute covering the investigated areas) were compiled and compared with on-site measurements (Fig. 1).

2.2. Water, sediment, and pore-water samples

Samples of water (n = 43 for each season) were collected based on their depth. Samples of surface sediments (n = 43 for each season) were collected to a depth of 5 cm by use of a grab corer. Locations of both water and sediment were collected and identified by use of a Magellan 315 GPS (Magellan, Taiwan). After homogenization, samples were stored in a portable refrigerator and then delivered immediately to the laboratory. To prevent oxidation, pore-water samples were separated from particulates in sediments by centrifugation (GT10-1, Beijing Era Beili Centrifuge Co., Beijing, China) in closed containers for 1 h at 3,000 rotations per minute. Concentrations of DTP, DOP, and TP were spectrophotometrically measured in the supernatant, using the molybdenum blue method [18–20].

The forms of P in sediments were operationally defined by using a scheme that consisted of five sequential extractions and characterized the forms of P as Fe/Al–P, Ca–P, OP, IP, and TP. The sequential extractions were concentrated HCl-extractable P (TP and total P), HCl-extractable P (Ca–P and P associated with Ca), NaOH-extractable P (Fe/Al–P, P bound to Al, Fe, as well as Mn-oxides and hydroxides), IP, and OP, were determined using the SMT protocol [21]. Glass- and plasticware were soaked in 0.3% HCl and rinsed with deionized water. Quality control procedures were applied throughout sampling, preparation, and analyses. For all samples, triplicate extractions with each extractant were performed and data were expressed as the average of these
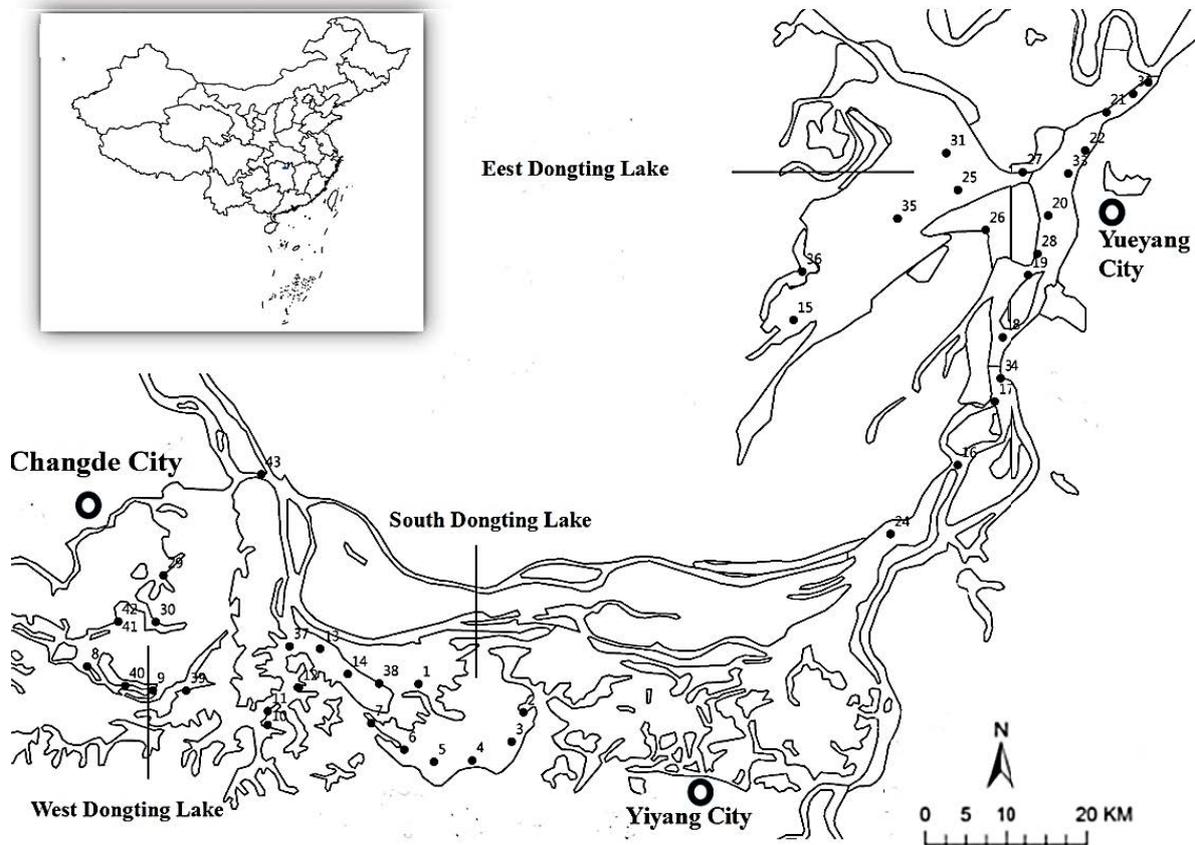
Fig. 1. Study area and sampling sites.

extractions. Relative deviations were expressed as standard errors and remained within 5%.

2.3. Statistical analyses

Statistical calculations were conducted using SPSS software (version 9.0). To analyze the differences in endogenous P from the bottom sediments under various environmental conditions and seasons, canonical variate analysis (CVA) was employed using Canoco for Windows 4.5 [22]. Release of P was used as descriptive variable. Supplementary environmental variables did not influence the ordinations of observed samples. Significances of particularly explanatory variables used in the model (DO, T, ORP, Cond., TDS, Turb., depth, and pH) were tested using a Monte Carlo permutation test, with 5,000 permutations. The correlation coefficient (r^2) was expressed as the product of the factor projection and cosine. To decrease the influence of spatiotemporal autocorrelation, a variable encoding particular sampling events was introduced for the analysis as a grouping of covariate defining blocks of data. Permutations were restricted to these blocks, if they formed a time series.

2.4. Model to predict the released amount of phosphorus

Considering the complexity and computational requirements of the direct incorporation of a numerical simulation

model into a further prediction framework to support process control, a statistical relationship between the predicted released amount of P and environmental factors (n = 43 samples) was established. For this, multivariable linear regression models (MLRM) were used. With the result of these MLRM, a predictive model was established based on environmental factors that could be easily and rapidly measured. In the past, the measurement method of P from sediments and the various forms of P were complicated and required a long time. However, it is possible to simulate these responses as a function of environmental factors that can be easily measured. Therefore, a linear regression equation for the prediction of P release was developed (Eq. (1)) [23]:

$$r = b_0 + \sum_{i=1}^{m} b_i x_i \tag{1}$$

Based on available monitoring information: the $x_{1n}, x_{2n}, \ldots,$ x_{nm}, r_n (n = 1, 2, …, n; n = 43 samples; where m represents the environmental factor number), set can be described (Eqs. (2)–(4)):

$$b_0 = \bar{r} - \sum_{i=1}^{m} b_i \bar{x_i} \tag{2}$$

$$x_i = \frac{1}{N} \sum_{i=1}^{m} x_{in} \tag{3}$$

$$\bar{r_i} = \frac{1}{N}\sum_{i=1}^{m} r_n \tag{4}$$

where i = 1, 2, …, m; b_0 and b_i represent regression coefficients (constants), r represents the concentration of TP in sediment (mg m^{-2}), and x_i represents the value of an environmental factor (e.g., T, pH, or ORP).

A set of linear equations were then solved simultaneously (Eqs. (5)–(7)) to identify the value of b_i [24,25]:

$$\begin{cases} L_{11}b_1 + L_{12}b_2 + \cdots + L_{1m}b_m = L_{1y} \\ L_{21}b_1 + L_{22}b_2 + \cdots + L_{2m}b_m = L_{2y} \\ \cdots \\ L_{m1}b_1 + L_{m2}b_2 + \cdots + L_{mm}b_m = L_{my} \end{cases} \tag{5}$$

where:

$$L_{ij} = \sum_{i=1}^{m}\sum_{k=1}^{n}\left(x_{ik} - \bar{x_i}\right)\left(x_{jk} - \bar{x_j}\right) \tag{6}$$

$$L_{iy} = \sum_{i=1}^{m}\sum_{k=1}^{n}\left(x_{ik} - \bar{x_i}\right)\left(y_k - \bar{y}\right) \tag{7}$$

Then, using the equation or SPSS software, the regression of r, which represents the concentration of TP in the sediment, can be obtained. The relationships among factors were assumed to be linear and exogenous inputs were assumed to be of short durations. SPSS 16.0 was used for the multivariate linear regression. Additionally, in the multiple linear regression, the degree of data fitting was assessed with the residual variance value S [26].

3. Results

3.1. Amounts and forms of P in sediment, pore water, and overlying water during wet and dry seasons

3.1.1. Sediment

Mean concentrations of TP in sediments followed the order of East Dongting Lake > West Dongting Lake > South Dongting Lake, with means of 712.5, 542.8, and 603.9 mg/kg, respectively (Table 1). The contents of TP and different forms of P in the three phases of sediment, pore water, and overlying water in different regions of Lake Dongting all changed. In the sediments, the IP concentration in the three areas of Lake Dongting was almost the same as that of OP, with an average of 40% of TP. However, the IP fraction, which included Fe/Al–P and Ca–P, differed among the three regions of Lake Dongting. In the East and West regions of the lake, the order could be ranked according to Fe/Al–P > Ca–P, which can frequently occur in eutrophic lakes [27]. This may be the result of inputs of industrial wastewater, domestic sewage, or non-point runoff from urban areas to the catchment of the lake [28,29]. The industrial development of the gross domestic product of Eastern and Western regions was 3.3 times higher than those of the Southern region. Dominance of Ca–P (with a mean of 36.9%) in the South region of Lake Dongting identifies the source of P as mainly non-point, that is, agricultural.

This was substantiated by the observation that this region also accumulated sand [30,31]. In general, the Fe/Al–P content is higher in areas with more pollution sources.

During the study period, eutrophication occurred in the Lake Dongting over the wet season. Comparison of the environmental factor data of each sample point (Table 2) showed that the environmental conditions in the wet season increased T, pH, and WD, while decreasing DO, ORP, and TDS. Furthermore, it can be speculated that the sediments released more P in these condition. Therefore, the release of P from sediments was further analyzed by measuring the TP variation of the Lake Dongting area (Fig. 2 and Table 1). Higher rates of release were observed in the Eastern and Western regions of Lake Dongting, with means of 1.7 and 1.0 mg/m^2 d, respectively. East Dongting Lake exhibited the highest rate of release. Rates of release were lowest in South Dongting Lake (with a mean of 0.3 mg/m^2 d). The order of relative proportions of P change form followed: Fe/Al–P > OP > IP > Ca–P. This indicates that Fe/Al–P and Ca–P can be released more easily from bottom sediments, which is consistent with observations in Lake Uzarzewskie, Poland, and Lake Tai, China [32,33].

3.1.2. Overlying water and pore water

The proportion of DTP and the form of dissolved inorganic P (DIP) always exceeded the proportion of dissolved organic P (DOP) in either the overlying water or pore water (Table 1 and Fig. 3). During the dry season, the mean concentrations of DIP and DOP in the overlying water body were 0.04 and 0.02 mg/L, respectively, while values of pore water were 0.02 and 0.01 mg/L, respectively. Concentrations of all forms of P in pore water were lower than those of overlying water. However, during the wet season, this condition was reversed because DIP in pore water was twofold higher, with a mean of 0.04 mg/L. However, the concentrations of DOP in pore water increased sixfold, with a mean of 0.07 mg/L. A significant increase in concentrations of P was found from dry season to wet season in pore water, which even exceeded concentrations in overlying water. Combined with Fe/Al–P changes in the same period, this could indicate that P was released from the bottom sediment to pore water, particularly Fe/Al–P. A concentration gradient was formed between pore water and overlying water. Then, dissolved P most likely transferred by concentration diffusion into the overlying water [34].

Based on the above results, and in reference to the change of Fe/Al–P in the sediment (Figs. 2–4), these results suggest that P was first released from the bottom sediment into the pore water (especially Fe/Al–P). A concentration gradient was formed between pore water and overlying water. Then, the DIP was most possibly gradually transferred to the overlying water through concentration diffusion.

3.2. Effect of environmental factors on P release

Experiments under laboratory conditions have been used to study the effects of various chemical–physical conditions in relation to P release [35–37]. However, P release is affected by the comprehensive action of many environmental factors. Comparing these P releases as a function
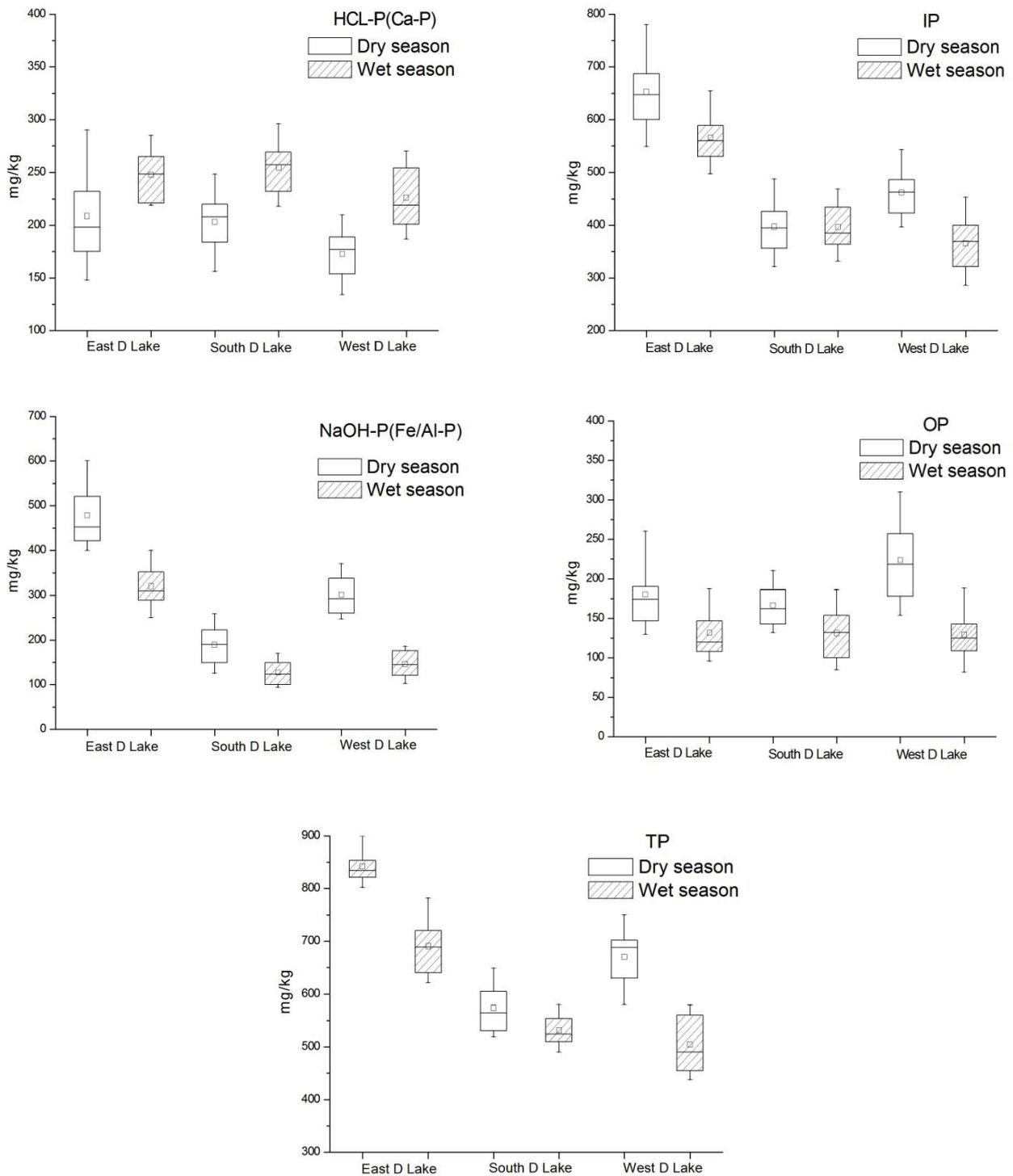
Fig. 2. Comparison of total concentrations of phosphorus (TP) released from sediments during wet and dry seasons.

of factors (Tables 1 and 2) showed that P release was associated with decreases in DO, ORP, conductivity, and TDS from 9.65 to 5.83, 168.30 to 113.56, 0.24 to 0.12, and 231.45 to 157.40, respectively. pH and T increased from 7.18 to 8.02 and 18.45 to 24.12, respectively. The environmental sequence diagram (Fig. 4) obtained by using CVA of 86 samples, further verified these parameters to be canonical factors. The diagram also depicts quantitative relationships. Greater variability of P release of was observed during the wet season, when the concentration of DO, as well as the ORP, pH, and depth was negatively correlated with rates of P release. This correlation followed the order

Table 1
Concentrations of total phosphorus (TP) in sediment, pore water and overlying water during the wet and dry seasons. Mean of triplicate samples, during wet or dry seasons; D: Dongtinghu)

Time	Samples	Sediment (mg/kg)					Overlying water (mg/L)			Pore water (mg/L)		
		TP	Fe/Al–P	Ca–P	OP	IP	DTP	DOP	DIP	DTP	DOP	DIP
	East D Lake	834.24	452.99	198.55	174.36	647.62	0.07	0.02	0.05	0.05	0.02	0.03
Dry season	South D Lake	564.83	190.91	208.42	162.11	395.61	0.04	0.01	0.02	0.02	0.01	0.01
	West D Lake	688.42	292.58	176.92	218.23	463.51	0.05	0.01	0.04	0.02	0.01	0.01
	East D Lake	689.28	310.96	248.19	120.13	560.22	0.09	0.03	0.06	0.12	0.08	0.04
Wet season	South D Lake	524.44	123.88	257.53	132.38	385.37	0.06	0.03	0.03	0.08	0.06	0.02
	West D Lake	490.27	145.72	219.15	125.05	369.11	0.07	0.02	0.05	0.10	0.07	0.03

of ORP > DOC > depth > conductivity ≈ TDS. The correlation coefficient was expressed as the product of the factor projection and cosine (r^2). At the sediment–water interface, DO controlled the redox potential and provided strong oxidation. Under aerobic conditions, P can be strongly bound with Fe^{3+} to form $Fe_2(PO_4)_3$. Therefore, P release from sediments is less likely under aerobic conditions. However, under anoxic conditions, sediments can more readily release P. Under anoxic conditions, P can only bind with Fe^{2+}, which has a much smaller solubility product [38]. That is why during the wet season, the level of Fe/P decreased and the results of CVA identified ORP as the driving factor for the release of endogenous P. Conductivity and concentrations of TDS are related to concentrations of ions, where P could settle in the sediment and bind with ferric ions [39]. Depth of the water column slightly affected DO and turbulence, which resulted in less resuspension at greater depths of the overlying water. During the wet season, pH increased (with a mean of 8) and release of P was simultaneously affected through the release of significant amounts of ORP. pH and temperature were positively correlated with P release (Fig. 4). pH affected the release of P by affecting the exchange of PO_4^{n-} [40]. Ion exchange between P, complexed with iron, aluminum, and OH^- at elevated pH results in release of P from sediment [41]. Temperature had a significant effect on the release of P, because temperature is also significantly correlated with reaction rate, diffusion rate, and other environmental factors [42] (Fig. 4).

3.3. Model to predict P release as a function of environmental factors

Based on r as derived above (Section 2.4), R_i, which is the rate of release of P can be derived (Eq. (8)). Then, based on the value of R_i, the dynamic rate of P release can be predicted by using a forecast model (SDPRFM) and Eq. (9).

$$R_i = \frac{r_i' - r_i}{\text{time}} \tag{8}$$

$$W = \sum R_i \cdot \Delta T_i \cdot A_i \tag{9}$$

where W represents the predicted mass of P released (t), R_i represents rate of P release under condition i (mg m^{-2} d^{-1}),

ΔT_i represents the time under condition i (d), A_i represents the lake area under condition i (m^2), r_i' represents the concentration of TP in sediment (mg m^{-2}) at a previous time step, and r_i represents the concentration of P in sediment (mg m^{-2}) at a later time step. A month was assumed to have 30 d.

Then, $R_i = 0.472 + 0.085[T] + 0.029[TDS] + 0.058[Cond.] - 0.207[ORP] - 0.134[DO] - 0.047[Depth]$. This helped to control the eutrophication of Lake Dongting. This formula was used to predict the possible P release (W_1 = 132 g) of East Dongting Lake over the next 3 months. Compared with the average P release (W = 78 g) of the recent 3 years, W_1 increased nearly twofold. This identifies a serious risk of P release in East Dongting Lake. This result was verified by a subsequent serious eutrophication event.

4. Discussion

4.1. Release mechanism of P forms

Redox reactions and physical diffusion of endogenous P at the sediment–water interface lead to various processes, such as desorption, dissolution, exchange during migration, and transformation of P [23,43] (Fig. 5). First, P was released from sediment to the pore water in interstitial spaces between the particles of consolidated bottom sediments. Then, most of the P diffused from the upper mixed soil interface into the overlying water body [44], while part of the P can be held in the sediment [45]. If rates of released P are high, P will accumulate in the overlying water. The upward exchange flux from pore water to the overlying water body is related to turbulence and resuspension of the sediment, and also depends on the following three processes [46]: (1) concentration-gradient dependent diffusion between pore water and overlying water body; (2) the redox reaction behavior of surface sediments, which potentially causes weak bonding of released P and; (3) release of P in both sediment and pore water because of ion exchange. All afore-mentioned environmental factors are important for determining whether these processes occur. The effects of environmental factors on the release of P and the reactions at the sediment–water interface are summarized in Fig. 5.

4.1.1. Sediment phase

The results of Section 3.1 showed that P release mainly occurs as Fe/Al–P. Decreases of DO and ORP result in the
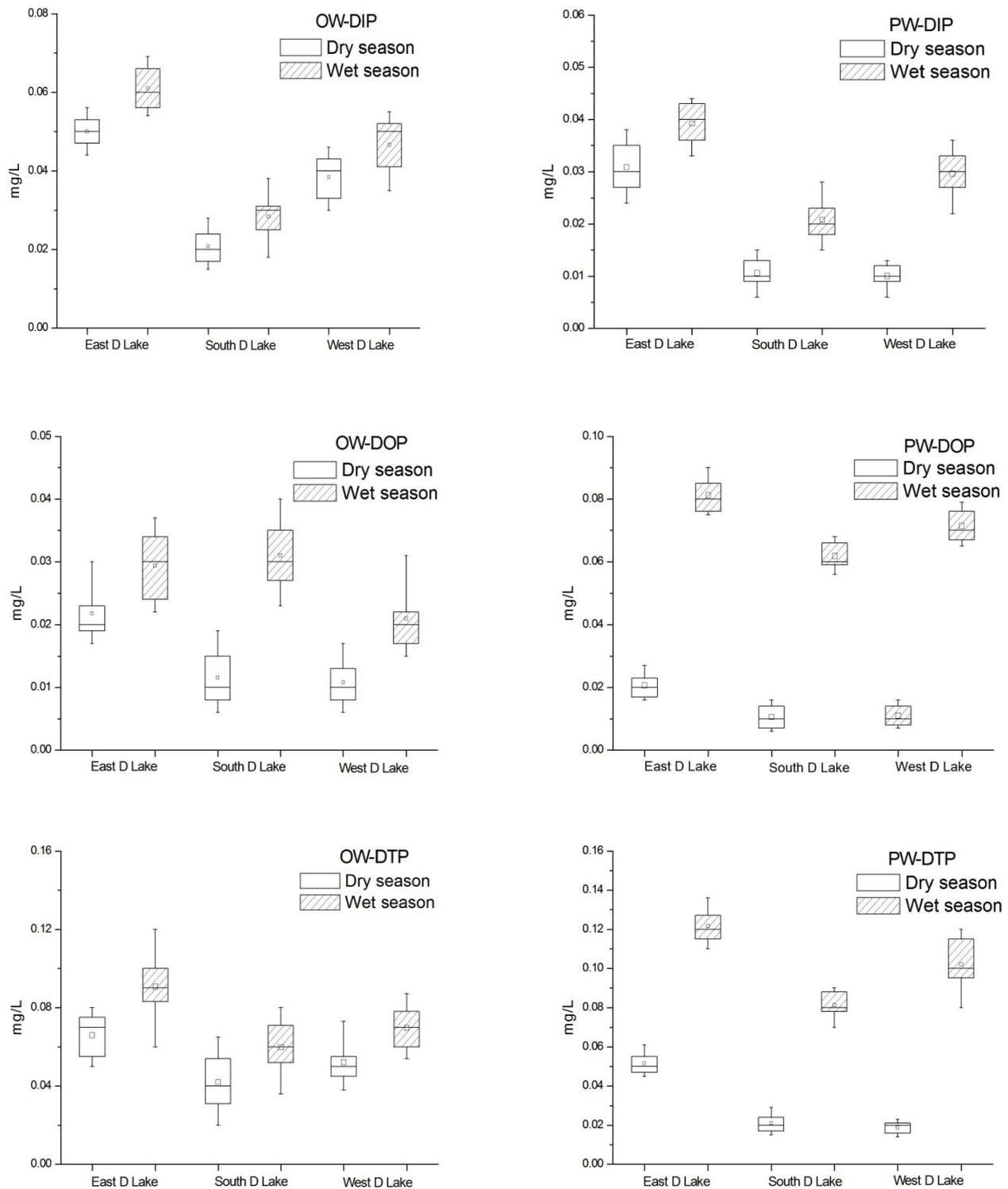
Fig. 3. Comparison of potential release of phosphorus (P) and proportions of P in overlying water and pore water, between dry and wet seasons (OW: overlying water; PW: pore water).

conversion of Fe^{3+} to Fe^{2+}, which weakens the affinity of PO_4^{3-}, combined with Fe/Al–P [47]. Also, decreases of pH result in increasing OH^- concentration. Ion exchange between P complexed with iron, aluminum, and OH^- results in

PO_4^{n+} release, which diffuses into the pore water (Eqs. (10) and (11)) [48]. Additionally, CVA showed that if the water depth is shallow, surface sediment is easily resuspended by hydrological conditions, which can cause direct P release into

Table 2
Environmental factors, measured 5 cm above bottom sediments (means of triplicates) during the wet and dry seasons

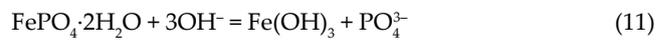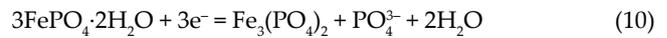| Time	Samples	T (°C)	DO (mg L^{-1})	PH	TDS	ORP	Cond.	Turb.	WD	Substrate
	East D Lake	19.46	9.57	6.94	257.00	118.80	0.26	102.33	2.73	Mud-sand
Dry season	South D Lake	18.33	10.23	7.23	225.00	173.90	0.23	100.56	1.67	Sand
	West D Lake	17.55	8.73	7.07	198.00	243.82	0.22	84.60	1.93	Hardcore
	East D Lake	25.28	6.01	8.05	170	79.20	0.15	93.24	5.47	Mud-sand
Wet season	South D Lake	23.80	6.12	8.04	150	125.11	0.11	93.27	2.78	Sand
	West D Lake	23.79	5.46	8.00	150	146.00	0.11	64.23	3.26	Hardcore
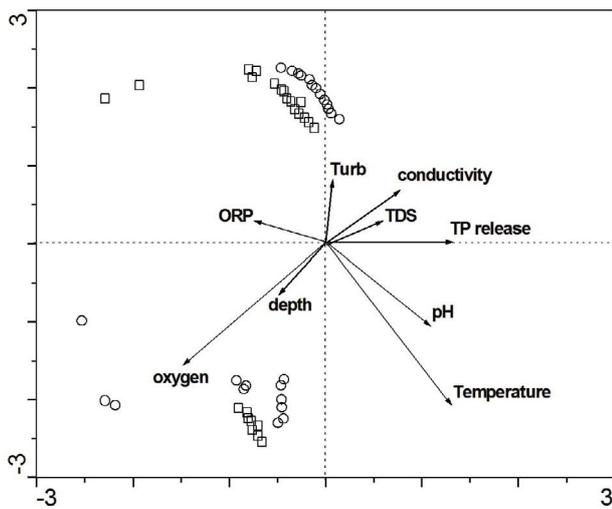
Fig. 4. Release of phosphorus (P) under various environmental conditions. Canonical variates analysis diagram showing release of P from bottom sediments, as a function of changes in environmental factors between two seasons (dashed lines – parameters that do not differ significantly; squares – dry season samples; circles – wet season samples).

pore water or overlying water body [49]. Increases in temperature slightly increase oxidation–reduction reactions and ion exchange, which can increase P concentrations [48].

$$3FePO_4 \cdot 2H_2O + 3e^- = Fe_3(PO_4)_2 + PO_4^{3-} + 2H_2O \qquad (10)$$

$$FePO_4 \cdot 2H_2O + 3OH^- = Fe(OH)_3 + PO_4^{3-} \qquad (11)$$

4.1.2. Pore water phase

The results of the changes of P content in pore water and overlying water body presented in Section 3.1 showed that the P content in pore water increased significantly, and even exceeding the concentration of P in the overlying water (Table 1). At this concentration gradient, P will be released from the pore water to the overlying water body via concentration diffusion. As mentioned in Section 3.2, the process is significantly affected by both temperature and pH. However, at the same time, under the influences of both higher concentrations of TDS and conductivity, mineral ions, such as iron, aluminum, magnesium, and calcium ions in the water can also combine with the precipitate formed by P in the pore water and then return to the sediment [50].

**P – Dissolved Phosphorus;   A – Ion Absorption;   B – Bioassimilation;
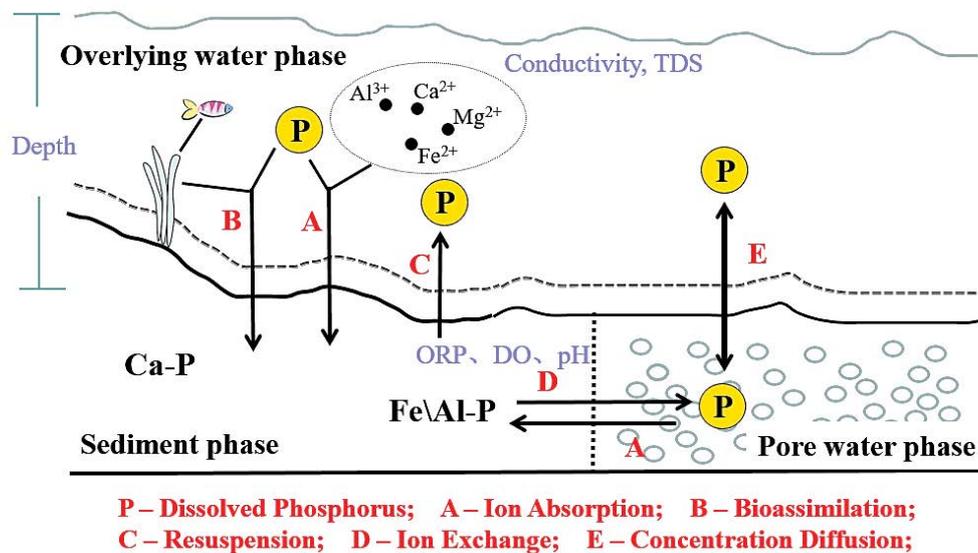C – Resuspension;   D – Ion Exchange;   E – Concentration Diffusion;**

Fig. 5. Schematic of endogenous phosphorus (P) migration and process for transformation at the sediment – water interface under various environmental conditions.

4.1.3. Overlying water phase

The TP in the overlying water increased significantly during the wet season, reaching up to 0.09 mg/L. Once the P migrated from the sediments from the pore water to the overlying water body, it can only be removed under the natural action of adsorption by mineral ions or via biological absorption [51]. Otherwise, a large amount of P will accumulate in the overlying water body, which will eventually cause cyanobacteria to multiply, thus leading to eutrophication.

4.2. Prediction of P releases with environmental factors

Controlling the eutrophication of lakes has been attempted for decades. However, it is very difficult to control eutrophication in time because of the complexity and variability of endogenous P release. This process involves changes of environmental factors, multiphase transformation, and release mechanism. Previous researchers constructed complicated white-box models (which focus on the input–output relationship in the control theory and reflect the processing status). Examples are the S-P model, the Thomas model, the Qual series model, and the WASP model, all of which explain transport and transformations at the sediment–water interface [52]. However, because the parameters used in such white-box models need to be tested over time, the concentrations of released P cannot be identified in real-time (Fig. 6). However, to control lake eutrophication, timely predictions of releases of endogenous P are needed [53,54]. These can be obtained by using monitoring data of environmental factors for a specific area to help develop strategies to remediate the area either via chemical, biological, or physical techniques [55]. This problem requires an urgent solution for the effective control of lake eutrophication. Thus, this study examined the effects of environmental factors on the release of P from the sediment. Significant quantitative relationships were identified between the release of P and changes in environmental factors. Hence, supports were desired to provide eutrophication control by constructing a model based on environmental factors to predict P releases from sediments.

These models assumed that relationships among factors were linear and that exogenous inputs have only short-term effects. A statistical relationship between the P release behavior and environmental factors (n = 43 samples) was established using MLRM analysis [56]. Through the results of this MLRM analysis and deductions of the predicted amounts of P released, a prediction model was obtained that is based on easily measurable environmental factors. This model can be used to predict the amounts of P released from sediments. The model, which is mainly based on Eqs. (1), (8), and (9), was used to predict the concentrations of released P from sediments under various combinations of conditions. In subsequent applications of this predictive model, large increases of released amount of P in the northwest bay of East Dongting Lake were found. At this location, eutrophication was indeed observed. At the same time, the simulation model suffered from a number of deficiencies, such as prerequisite for specific assumptions, other possible influencing factors, and coefficient modification. This requires more data and detailed derivations.

5. Conclusions

An analysis of the released endogenous P across the sediment–water interface in Lake Dongting showed that the migration of endogenous P from the sediment to the overlying water body follows a three-phase cycle process. Iron and aluminum phosphorus (Fe/Al–P), calcium phosphorus (Ca–P), and total P transformation will occur. Fe/Al–P is the most important source of the released P. The content of total P in Lake Dongting follows the order of East Dongting Lake > West Dongting Lake > South Dongting Lake. The contents of Fe/Al–P were highest in East Dongting Lake, which may indicate a risk of P release. CVA showed that the effect relationship on P release follows ORP > pH > T > depth > conductivity ≈ TDS. Among these, ORP was the most important driving factor, as it affects the conversion of Fe^{3+}–P and Fe^{2+}–P. The pH value affects ion exchange between PO_4^{n+} and OH^-, and depth is related
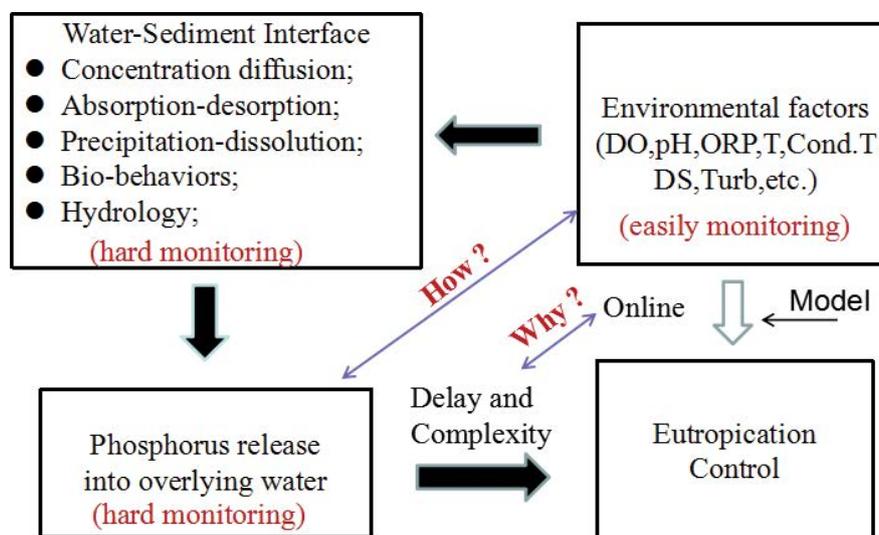

Fig. 6. Schematic showing how and why to build a predictive model based on environmental factors.

to disturbance and resuspension. All of these effects cause direct releases of P into the pore water. Finally, a model to predict P release was developed based on environmental factors that can be easily measured. The P release rate (R_i) was $R_i = 0.472 + 0.085[T] + 0.029[TDS] + 0.058[Cond.] - 0.207[ORP] - 0.134[DO] - 0.047[Depth]$. If employed appropriately in eutrophication engineering, this predictive model constitutes an important step toward better sustainability, easier integration with engineering concepts, and improved risk management. Moreover, as previously noted, this prediction requires far more data and detailed derivations (such as subdividing depth into flow velocity and wind frequency).

Acknowledgments

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