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# Contrasting effects and mode of dredging and *in situ* adsorbent amendment for the control of sediment internal phosphorus loading in eutrophic lakes

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ABSTRACT

Dredging and *in situ* adsorbent inactivation are two methods which are frequently used in eutrophic water bodies such as ponds, lakes and estuaries to control internal phosphorus (P) loading from sediments. However, their effects and modes on the control of sediment P loading has been seldom compared. In this study, a long-term sediment core incubation experiment in the field was undertaken to investigate changes in sediment P loading (P fluxes, supply ability and forms of P and transformation) comparing two remediation techniques, that of lanthanum-modified bentonite (LMB) addition or dredging to a control. A 360-day field investigation indicated that LMB addition more effectively reduced pore water P concentrations and sediment P fluxes than dredging in comparison with the control. On average, dredging and *in situ* LMB inactivation reduced the P flux by 82% and 90%, respectively relative to the control sediment. Whilst both the LMB inactivation and dredging can reduce the mobile P concentration, the impact of LMB in reducing mobile P was demonstrated to be more prolonged than that of dredging after 360 days. The P fraction composition in the LMB inactivated sediment differed significantly from the dredged and control sediment. Contrary to physical removal of dredging, chemical transformation of sediment mobile P and Al-P into Ca-P is the main function mode of LMB for sediment internal P control. Both LMB addition and dredging caused changes in the composition of sediment bacterial communities. Whilst LMB addition increased bacterial diversity, dredging temporarily reduced it. This study indicates that *in situ* inactivation by LMB is superior to dredging in the long-term control of sediment P loading.



## 1. Introduction

There is international consensus that internal phosphorus (P) loading from lake bottom sediments will contribute to a decline in lake water quality even if external P loading can be effectively reduced (Marsden, 1989; Schindler et al., 2016; Schönach et al., 2018). Critically, P can be periodically released from sediment to overlying water underpinning eutrophication and algal growth (Jeppesen et al., 2005; Søndergaard et al., 2013; Douglas et al., 2016). In one example it was estimated that more than 90% of P in eutrophic lake was stored in bottom sediment (Søndergaard et al., 2013). Therefore, inhibition or effective interception of P released from bottom sediment will constitute amongst the most important management actions in a eutrophic lake once the external input of P is limited.

Sediment dredging and *in situ* adsorbent amendment are the two most commonly used options to control of P release from bottom sediment (Lürling and van Oosterhout, 2013). Dredging surface sediment that contains P in addition to some forms of N and other pollutants such as trace metals can likely reduce the internal P load (Oldenborg and Steinman, 2019) via a reduction of bottom sediment fluxes of P into the overlying water (Chen et al., 2018a). Nonetheless, there remains considerable debate with respect to the efficacy of sediment dredging. Several studies indicated that sediment dredging can effectively reduce the release of P and metals from sediment (Yu et al., 2019; Li et al., 2020). On the contrary, other studies indicated that sediment dredging does not constitute a long-term management option for sediment internal P loading especially whilst external loading continues (Kiani et al., 2020; Wen et al., 2020) with effective P retention augmenting the

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bottom sediment inventory. Secondly, the disturbance of sediment may cause the release of co-contaminants such as heavy metals and therefore induce secondary pollution (Yu et al., 2019). Moreover, sediment dredging can produce a large quantity of waste which will require dewatering and land disposal. In many cases the dredged sediments may be reactive once dried requiring further management. However, dredging is still used in the management of sediment-associated internal loading around the world based on the premise that sediment dredging can decrease endogenous P load in a lake and therefore assist in the mitigation of eutrophication.

*In situ* P adsorbent amendment is another method which is frequently used in eutrophic lakes for management of sediment-associated internal P loading (Robb et al., 2003; Haghseresht et al., 2009; Spears et al., 2013; Yin et al., 2018; D'Haese et al., 2019; Douglas et al., 2019). Adsorbents which contain chemical inactivation agents such as La or Al can bind mobile and labile P in sediment into inert forms (Lürling et al., 2016), effectively reducing P flux from sediment. Amongst the most common of P inactivation methods invented is lanthanum-modified bentonite (LMB), known commercially as Phoslock, which has been used in hundreds of lake and reservoir trials and full-scale applications for eutrophication control (Copetti et al., 2016). Many results have documented that the addition of LMB can effectively lower down the P concentrations in overlying water immediately following application while also providing a reactive barrier to intercept internal P loading derived from the bottom sediment (Mucci et al., 2020). The basis of LMB-based treatment is the formation of rhabdophane ($LaPO_4 \bullet nH_2O$), a precipitate resistant to redox changes and stable over a wide pH range (Slade and Gates, 1999; Meis et al., 2013), although depending on the composition of the water, a range of intermediate phases may be present (Mucci et al., 2020).

Despite the demonstrated efficacy of P inactivation agents such as LMB, there is controversy on the long-term effectiveness on P control by these chemical inactivation agents in shallow lakes (Lin et al., 2019). This arises in cases where there may be frequent sediment resuspension, bioturbation and the deposition of suspended particulate matter. All of these processes can result in burial, dispersion or preferential focusing of the applied materials in the bottom sediment (Xu et al., 2017), with the potential to compromise the effectiveness of P-inactivation agents.

The selection of the most appropriate methods (dredging and *in situ* sediment inactivation or a combined approach) in dealing with the remediation of sediment P load is a critical step for the effective management of water bodies. This selection, often with a substantial associated cost (Douglas et al., 2016), will to some extent determine the success of lake restoration. Currently, few studies have been undertaken to compare the two methods in the efficiency of sediment P loading management. Therefore, a simulated field sediment core experiment was undertaken to investigate the effectiveness of dredging and *in situ* inactivation on the reduction of sediment P loading (including porewater P concentration, P flux and P forms transformation). Furthermore, the changes in bacterial community in the dredged and inactivated sediment were also analyzed and compared to the changes in sediment P fractions. The results of this study will inform the selection of methods for the management of internal sediment P loading in eutrophic lakes.

## 2. Material and method

### 2.1. Materials

Lanthanum-modified bentonite (Phoslock) was obtained from Beijing Fusileke Water Solutions Ltd and was used without further modification. Intact sediment cores (no less than 50 cm long) were collected from Western Lake Chaohu (117.3489° N; 31.687° E) using core sampler 8.8 cm in diameter and 60 cm in length on January 2019. The site was selected due to the known high sediment P and N internal loading based on previous study (Yang et al., 2020). In total, 70 sediment cores were collected for the experiments. All of the sediment cores were temporarily brought to shore for handing. Approximately 100 L water was collected from the sampling site for the experiments.

### 2.2. Experimental design

A total of 54 sediment cores were randomly selected, and placed into experimental tubes, followed by filling with lake water. A total of 18 sediment cores were used for each treatment: (1) Control, where the sediment cores were intact and untreated; (2) Dredging (Dred), sediment cores with the upper 25 cm surface sediment was sectioned and removed to simulate sediment dredging (3) Lanthanum modified bentonite (LMB) sediment cores with LMB added in a ratio of 100:1 based on the mass of mobile P where the amount of mobile P was estimated from that present in surface sediment (0–5 cm) in the sediment cores and the soluble reactive phosphorus (SRP) in the overlying water.

To avoid the deposition of suspended particulate matter into the sediment cores, all sediment tubes were covered with polyamide screens with an 800 mesh (18-$\mu$m mesh size) and which was periodically changed. All sediment tubes were then placed into the field observation platform (31°42′6.53″N; 117°23′19.23″E) with submersible, stainless steel frames to house the cores and designed to simulate actual lake environmental conditions. The three groups of treated sediment cores were placed in stainless-steel frames and submerged at the bottom of the lake and tethered with a nylon rope connected to the platform. The sediment tubes were retrieved after 30 (Feb, 2019), 60 (March 2019), 90 (April 2019), 180 (July 2019), 270 (October 2019) and 360 days (January 2020). At each sampling time, nine sediment tubes (three sediment tubes for each treatment) were collected and used to measure P flux and sediment P fractions and analysis for bacterial community composition. Following the last sampling, diffusive gradients in thin film (DGT) measurements were performed with detailed information mentioned below.

### 2.3. Pore water sampling, p fluxes and diffusive gradients in thin film (DGT) measurements in inactivated and dredged sediment

The P fluxes across the sediment-water interface were measured by a high-resolution peeper (HR-peeper) sampling method at each sampling time. Detailed information on the use of HR-Peeper can be found in the supplementary material. The P concentration in sediment pore water in the HR-peeper was measured and used for calculation of the P fluxes across sediment-water interface.

The $PO_4^{3-}$ fluxes ($J$) across the sediment-water interface from the control and treated sediments were estimated from the pore water profiles of $PO_4^{3-}$, using Fick's first law of diffusion:

$$J = -\varphi D_s \left( \frac{\delta c}{\delta x} \right) (x = 0) \tag{1}$$

where $\varphi$ is porosity, $D_s$ is the bulk sediment diffusion coefficient, and $(\delta_C/\delta_x)_{(x\ =\ 0)}$ is the pore water concentration gradient across the sediment–water interface. $D_s$ was calculated from the diffusion coefficient of $H_2PO_4^-$ in water using $\varphi^3$ for $\varphi \geq 0.7$.

Two days before the end of the final sediment core incubations, commercial Zr-oxide diffusive gradients in thin films (Zr-oxide DGT) probes (Easy sensor Ltd, www.easysensor.net) were used to measure the two-dimensional (2D) labile-P for each treatment. The DGT probes were inserted vertically into the three groups of sediment cores and retrieved 24 hrs after deployment. The high-resolution 2D labile P determination and analytical methods are described in detail by Ding et al.(Ding et al., 2015).

### 2.4. Phosphorus fraction and sediment properties analysis

Phosphorus fractions in the sediment were determined using the sequential extraction method developed of (Rydin, 2000) as described in Supplementary Material according to (Zhang et al., 2014) with some modifications. In brief, air-dried sediment P was fractioned using $NH_4Cl$-P (loosely sorbed-P), BD-P (Fe-P), NaOH-nrP (Org-P), NaOH-rP (Al-P), HCl-P (Ca-P) and Residual-P (Res-P). The P concentration in each step was analyzed using the molybdenum blue method and expressed as an average of the two duplicate analyses. The total phosphorus (TP) in sediment was estimated as the sum of the six P fractions. The water content was measured by drying sediment samples at 105 °C for 24 hrs to a constant weight, and organic content using the Loss on Ignition Method (LOI, 550 °Cfor 4 hrs).

In order to study the composition of the benthic microbial community, around 1.0 g surface sediment (0–2 cm) from various treatments at different sampling times was sampled for DNA extraction. The DNA was extracted with a soil DNA isolation kit (MO BIO Laboratories, Carlsbad, CA, USA), following the manufacturer's instructions and then sequenced in an Illumina® MiSeq® platform (Personal Biotechnology Company, Shanghai, China). Detailed information on sequencing and data analysis is provided in the Supplementary Material.

### 2.5. Statistical analysis

Significant differences in variables (P flux, P fractions and pore water P concentration) among treatments were identified through one-way analysis of variance (ANOVA) followed by Tukey's test. Typically, homogeneity of variance for the obtained data was tested and data of the test values > 0.05 were adopted for the ANOVA analysis. All statistical analyses were performed using SPSS 19.0 (IBM, New York, USA), with significant levels $p < 0.05$ and $p < 0.01$ reported. Correlations between bacteria community structure and environmental variables were explored with the linear-model-based redundancy analysis (RDA) using the software Canoco for Windows (version 4.5).

## 3. Results and discussion

### 3.1. Sediment properties

The nutrient concentrations in the sediment used in this study are listed in Table S1. The average concentrations of total P (TP) and total N (TN) in the cores were 1003 mg/kg and 1938 mg/kg, respectively. These TP and TN were 1.6 and 1.2 times of that of average of TP and TN in Lake Chaohu, respectively (Yang et al., 2020) confirming that the site chosen for this study was likely to constitute a region of higher internal nutrient loading. The high concentrations of TP and TN confirm that this area has received long-term pollution load from the Naifei River which flows through the urban area of Hefei City. The mobile P fraction (the sum of $NH_4Cl$-P, BD-P and NaOH-nrP) was estimated to be 395 mg/kg or 39% of the TP. The loss on ignition (LOI) of the sediment was 6.5% as indicated Table S1.

### 3.2. Pore water p concentration, fluxes and p supply capacity from dredged and inactivated sediment

The temporal variation of sediment pore water P concentrations along sediment depth from the three treatments are presented in Fig. 1. The results indicated that the pore water SRP concentration in the control cores increased with temperature reflecting seasonal variations (Fig. S1). Maximum SRP concentrations in the control sediment typically occurred at 2–3 cm below the sediment-water interface and then decreased steadily with sediment depth. In comparison, porewater SRP concentrations in the dredged and LMB-treated sediment showed considerably less variation in the profiles as a function of depth with many of the profiles often indistinguishable within one standard deviation (Fig. 1). A seasonal variation of porewater SRP in both concentration and depth from control sediments has been reported in previous studies (Markovic et al., 2019; Yang et al., 2020). Higher temperatures will often accelerate microbially-mediated degradation of organic matter in sediment, which may increase SRP concentrations in sediment pore water (James, 2017; Chen et al., 2018b; van Dael et al., 2020).

Both dredging and LMB inactivation can effectively reduce the P concentration in sediment porewater and this is especially evident in warmer seasons (July 2019 and October 2019, Fig. 1). For example, dredging and LMB inactivation reduced average porewater SRP concentrations at day 270 by 60% and 69%, respectively when compared with the control sediment. This outcome is important for the management of eutrophic lake sediment internal P loading, particularly where higher sediment nutrient fluxes can be observed during the summer where algal blooms may occur (Markovic et al., 2019; Yang et al., 2020). The strong reduction of sediment pore water P concentrations, by the addition of LMB in particular, will also reduce the magnitude of P release from sediment.

Estimated P fluxes across sediment-water interface from the three treatments are presented in Fig. 2. The of P flux from the three sediment treatments reflected sediment pore water P concentrations (Fig. 1) with higher P fluxes observed in warmer seasons and reduced fluxes at lower temperatures (Gibbons and Bridgeman, 2020). The control sediment P fluxes gradually increased from 0.27 mg/m$^2$.d in Feb 2019 to 4.12 mg/m$^2$.d in Oct 2019 and then decreased to 0.24 mg/m$^2$.d in January 2020. In comparison, sediment from dredging and LMB treatments maintained relatively low P fluxes with reductions of 80% and 92%, respectively compared with the control sediment in October 2019. On average, dredging and *in situ* LMB inactivation reduced the P flux by 82% and 90%, respectively relative to the control sediment. Importantly, LMB always performed better in reducing the sediment P flux than sediment dredging over the 360 days of field incubation.

To further investigate the long-term control effects of different engineering methods on sediment P loading, sediment labile P was measured using a high-resolution 2D method at the end of the experiment (Fig. 3). The 2D image of sediment labile P reflected the effects of dredging and *in situ* inactivation by LMB. High DGT labile P concentration or fluxes were observed in intermediate and deeper sediment horizons in control and dredged sediments, respectively. Furthermore, a number of more intense "hot spot" sites were observed for the control relative to the LMB inactivated sediment. In contrast, a deeper zone of sediment labile P was observed in the −60 to −80 cm depth interval of the dredged core (Fig. 3). Also noteworthy is a zone of low sediment labile P around the margins of the LMB inactivated core which may reflect downward migration, potentially controlled by bioturbation and wind–induced redistribution of this material following application.

Previous studies have indicated that the LMB can migrate into deeper sediment via bioturbation (Ding et al., 2018b; van Oosterhout et al., 2020). Therefore, it is likely that any migrated LMB in deeper sediment can react with mobile P in sediment and thus reduce its ability to diffuse into pore water and/or into the overlying water column (Dithmer et al., 2016). In contrast, it is likely that the mobility of P from the dredged sediment may increase, often due to the deposition of new P-rich sediment, such that the remedial effects of dredging will decrease over time.

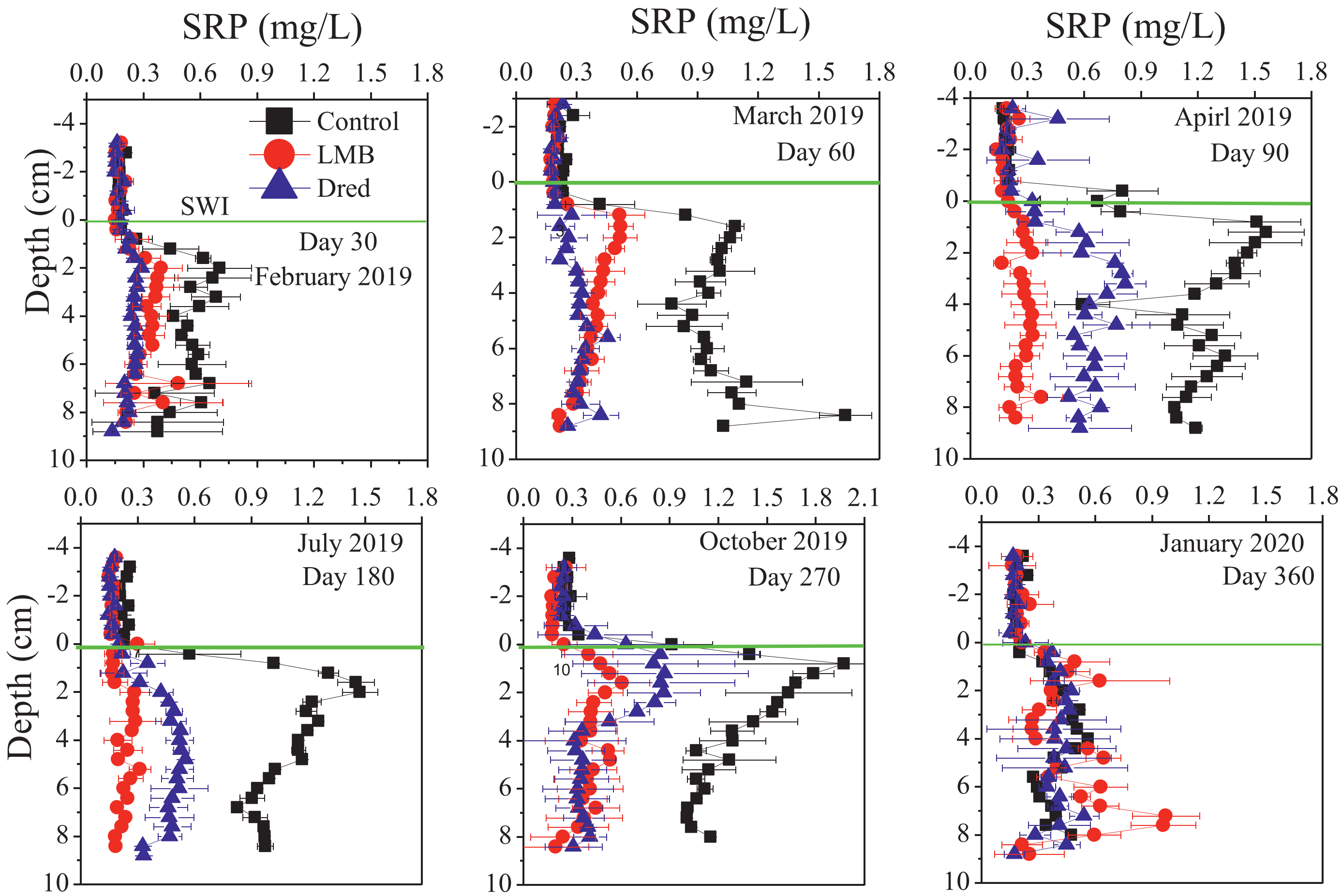


**Fig. 1.** Pore water soluble reactive phosphorus (SRP) concentrations for control, LMB inactivated and dredged sediment at six sampling times.

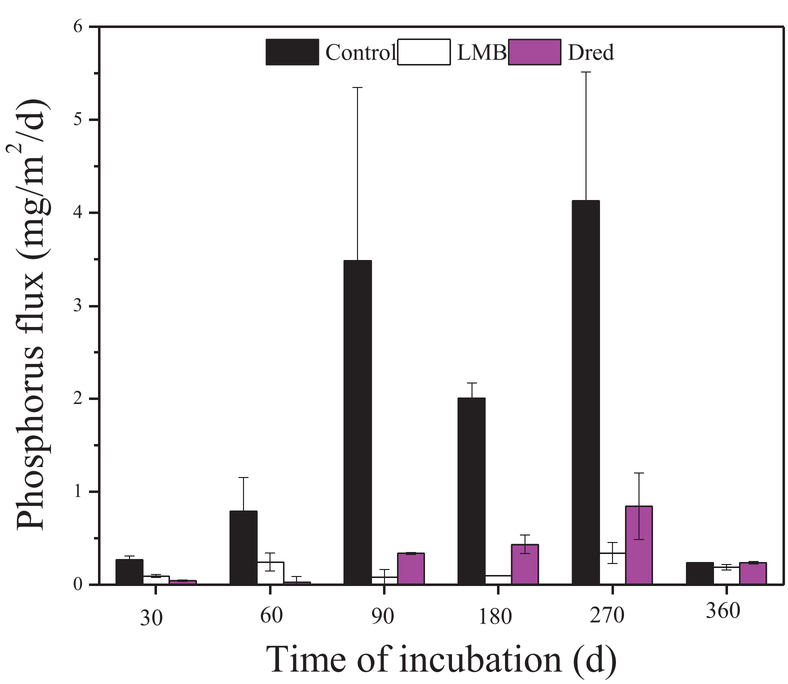


**Fig. 2.** Phosphorus fluxes across sediment-water interface for control, LMB inactivated and dredged sediment treatment at six sampling times.

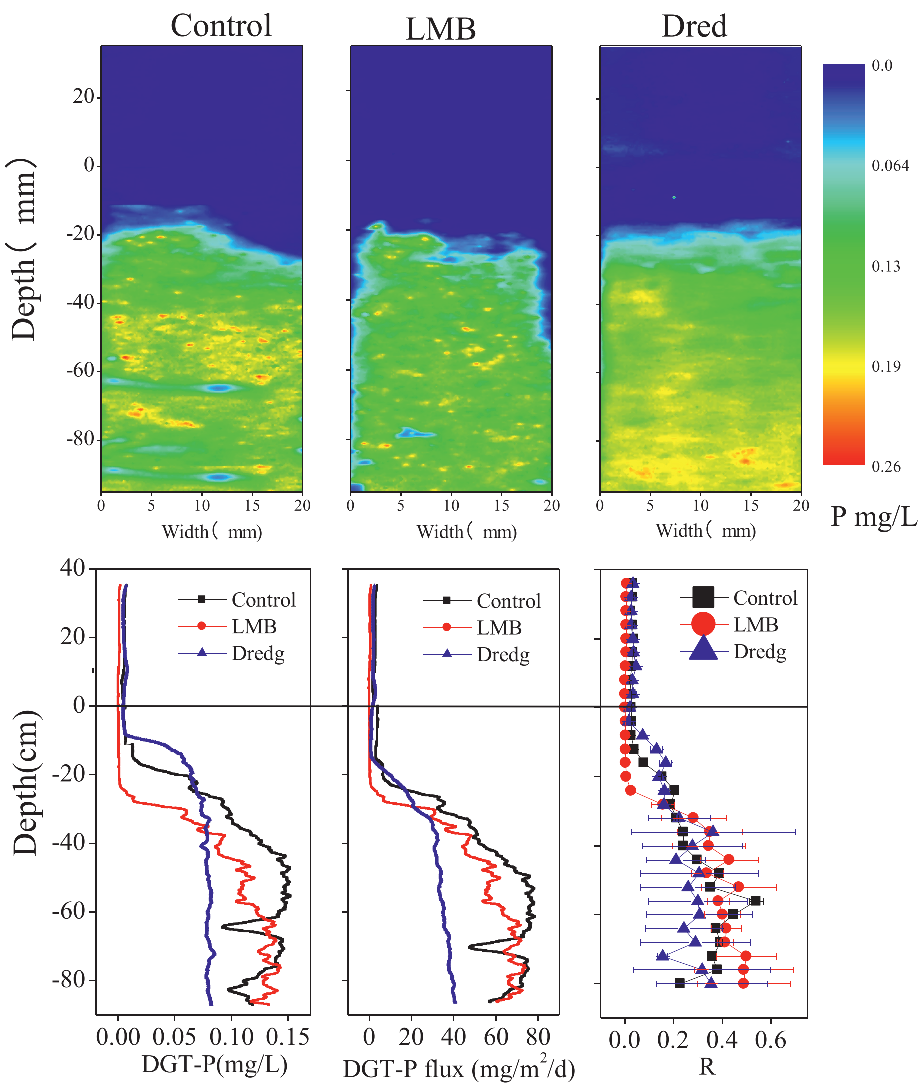


**Fig. 3.** Two-dimensional diffusive gradient in thin film (2D-DGT) phosphorus , 1D-DGT and R value profiles for control, LMB inactivated and dredged sediment treatment as measured 2 days before the end of the incubation experiment.

A calculated R value, expressed as the ratio of DGT labile P contained in the sediment relative to sediment pore water P, is also shown in Fig. 3. The DGT-measured fractions come from pore water and the further release of the solute from the sediment solids to resupply the pore water (Zhang et al., 1995). The lower of R value, the more difficult P diffusion from sediment to pore water is likely to be (Zhang et al., 1998). The average R value from the LMB inactivated sediment was only 0.007, considerably lower than the R value from the control and dredged sediment of 0.44 and 0.34, respectively. The R values of the LMB inactivated sediments were particularly low from 0 to −30 cm depth where the bulk of the sediment P-flux was likely to originate. This was in accordance with the estimated overall P flux (Fig. 2). The changes of sediment pore water P concentration, P fluxes and DGT measured P indicated that the LMB inactivation provides a potentially improved remediation outcome in terms of managing the sediment P-flux via P-binding relative to dredging where only the sediment is removed.

### *3.3. P forms in dredged and lmb inactivated sediment*

The surface sediment P fractions in the three treatments during various incubation times are shown in Fig. 4. All P fractions were extracted from the air-dried sediment as described in Section 2.4. These results may be different from the P fractions extracted using wet sediment as indicated in previous studies (Zhang et al., 2014). While this study has chosen to use air-dried sediments in P-fractionation experiments (Hupfer et al., 1995; Ruban et al., 1999; Kumar and Anshumali, 2019), a diversity of approaches exist including the use of freeze-dried sediments (Ruttenberg, 1992; Meis et al., 2012b; Kurek et al., 2020) in addition to wet sediments (Rydin, 2000; Huser and Pilgrim, 2014). It is understood that changes in operationally-defined P fractions may occur due to drying. The outcomes in this study have only been used to give a general indication of the predominant P fractions in the sediment. Critically, as will be presented later, we have demonstrated major changes in the conversion of a number of P fractions following imposition if the two remediation techniques. Furthermore, we have demonstrated that there has been a major change into an La-P bound fraction following the application of LMB that would be unlikely to be dependent on whether wet or dry sediments were used in the P-fractionation analysis.

The results indicate that the P compositions in the control, dredged and inactivated sediments were substantially different and showed dynamic changes during seasonal, and associated temperature variation. In control sediment, Al-P (NaOH-rP) was the main component of the total phosphorus (41–66% of sediment TP). This

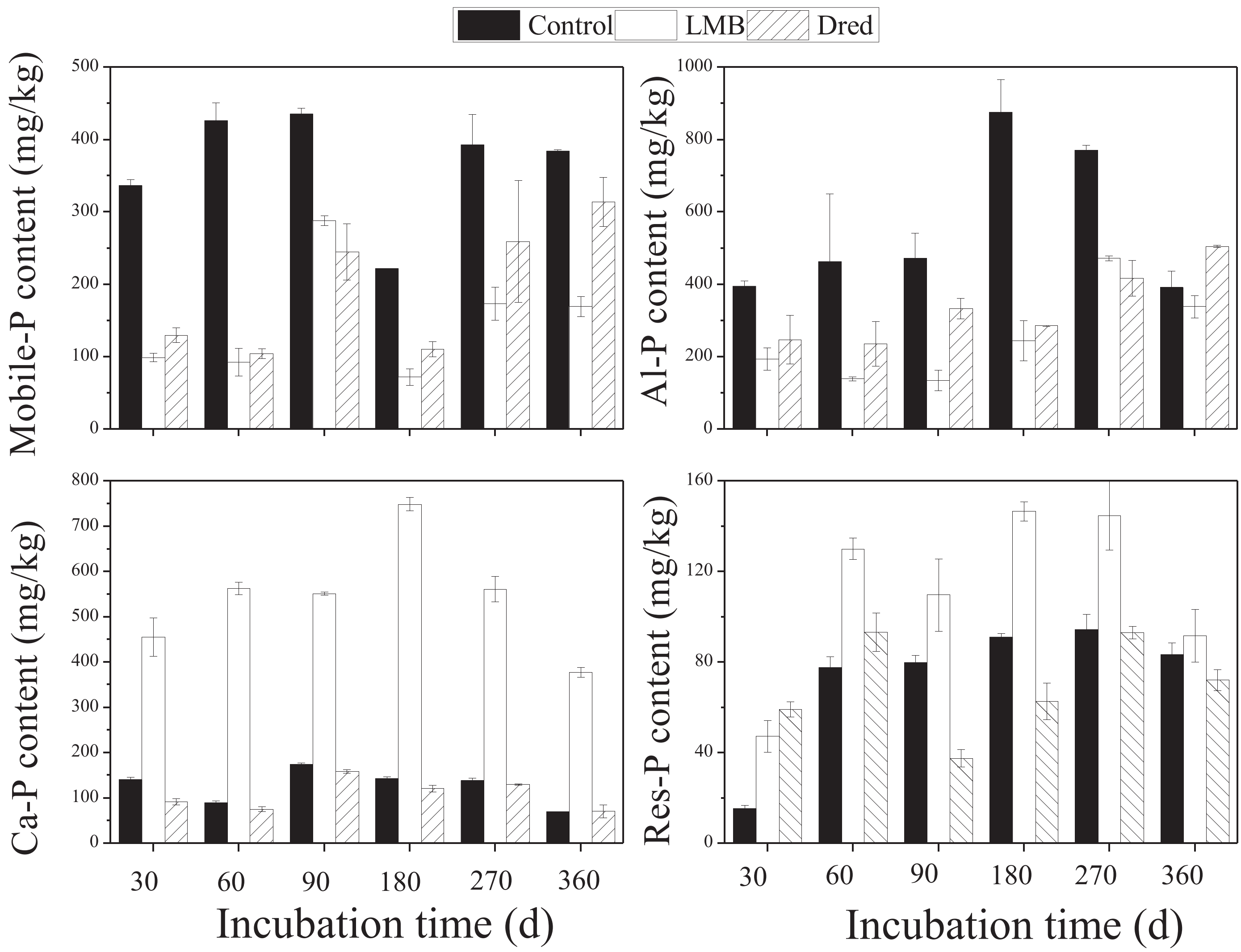


**Fig. 4.** P fractions in control, LMB inactivated and dredged sediment treatment at six sampling times.

is consistent with a previous study which indicated that Al-P predominated in Western Lake Chaohu bottom sediment (Yang et al., 2020). The mobile P concentration (the sum of labile-P, iron bound P and organic fractions of P) was in the range of 222 to 435 mg/kg with an average concentration of 366 mg/kg during the study. The mobile P fraction accounted for an average of 33% of the TP. The mobile P is regarded to be the chemically-defined fraction responsible for the active sediment internal P loading in eutrophic lakes. This is because this fraction of P has the potential to be released from sediment under anoxic conditions or as a result of microbial action (Reitzel et al., 2005). Hence, the reduction of this concentration of mobile P is considered an effective method to reduce lake sediment internal P loading (Funes et al., 2016).

The relative proportions of P fractions from the LMB inactivated sediment was markedly different to that of the control and dredged sediment. The mobile P concentration in the dredged and LMB inactivated sediment were in the range of 104 to 313 mg/kg (average 194 mg/kg) and 72 to 173 mg/kg (average149 mg/kg), respectively. The results indicate that both sediment dredging and LMB addition can reduce the concentration of mobile P in sediment, albeit by two distinct modes. The mobile P concentration in the dredged sediment was lower than that of the mobile P concentration in control sediment. However, mobile P accounted for 19 to 33% (average of 26%) of TP in dredged sediment which is similar to that of the percentage of mobile P in control sediment (17–41%, average 34%) (Fig. 5). In contrast, the addition of LMB largely reduced the percentage of mobile P in sediment which was in the range of 6 to 17% (average of 14%). Thus, on average, LMB addition reduced more than 60% of mobile P relative to that of the control sediment.

The nominally inert Al-P fraction in sediment was also significantly reduced after sediment dredging and LMB addition. The lower Al-P fraction in dredged sediment was due to the Al-P fraction in natural sediment generally decreasing with sediment depth (Li et al., 2020). However, the Al-P fraction was still dominant in the dredged sediment (average of 47%). In contrast, the Al-P fraction in the LMB inactivated sediment was on average 24% of sediment TP. The reduced sediment Al-P fraction seems to have been transformed into stable La-bound P (assumed to be contained in the Ca-P fraction) with this fraction increased significantly relative to the control and dredged sediment ($p<0.01$). On average, the Ca-P fraction in the LMB inactivated sediment was 4.6 and 5.2 times of that of the Ca-P fraction in the control and dredged sediment, respectively. This is consistent with many previous studies which reported that the LMB addition can increase the chemically-defined Ca-P fraction in sediment (Meis et al., 2012a; Li et al., 2019). The transformation of mobile P in sediment into a stable, nominally Ca-P bound fraction, will likely be beneficial to the long-

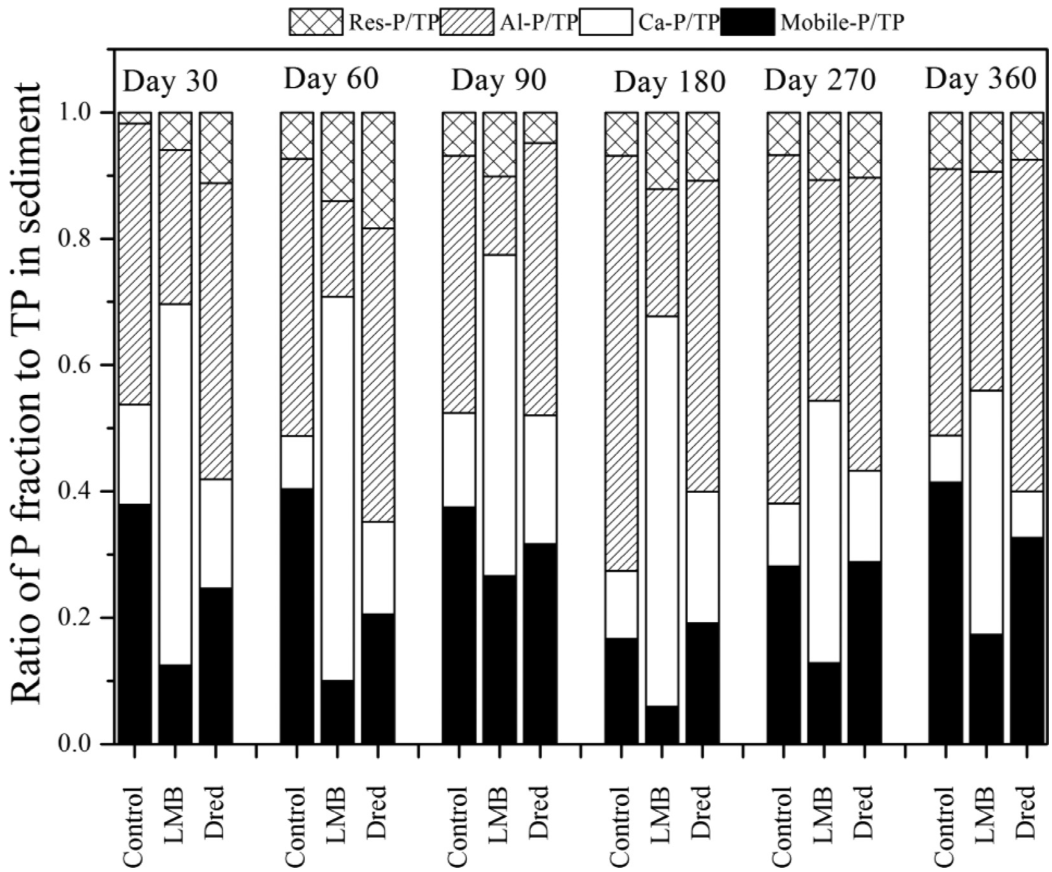


**Fig. 5.** The ratio of various P fractions to total phosphorus (TP) in sediment from control, LMB inactivated and dredged sediment treatment at six sampling times.

term management of sediment P in eutrophic lakes (Epe et al., 2017). This is because the P bound to LMB is considered stable and thus, will likely reduce the extent of eutrophication arising from internal P loading (Nürnberg, 2017).

### 3.4. Bacterial community in dredged and inactivated sediment and the relationship with sediment p fractions

The bacterial community diversity in sediment from different treatments and sampling times is listed in Table S3. The results indicated that the sequencing depth was enough because the coverage of the samples was more than 94% which means that most of the bacterial OTUs in each sample were captured. The results indicated that the remediation method can influence the bacterial diversity of the sediment. For the first four sampling times, the bacterial diversity indexes such as Chao l, Ace and Shannon in LMB amended sediment were higher than those for the control sediment. On the other hand, for the first two and last two sampling times, the values of Chao l, ACE and Shannon indexes for the dredged sediment were lower than those for the control sediment. These results indicated that the addition of LMB may temporarily increase the bacterial community diversity in the bottom sediment. The LMB can form a 2–3 mm thin-layer across sediment-water interface (Song et al., 2020). This may create a new type of microenvironment across sediment-water interface and hence favor the growth of different bacteria in the surface sediment. On the other hand, dredging can remove most of the upper organic and nutrient-rich sediment and also associated bacteria. However, the diversity of sediment bacteria can recover over time as observed in this study.

The relative abundance of various bacterial taxa at phylum level in the sediments from different treatments at six sampling times are presented in Fig. 6a. The most dominant phyla were *Proteobacteria, Chloroflexi, Bacteroidetes, Nitrospirae, Acidobacteria,* and *Patescibacteria*. Other phyla included *Actinobacteria, Verrucomicrobia, Epsilonbacteraeota, Planctomycetes, Omnitrophicaeota, Firmicutes, Gemmatimonadetes, Spirochaetes, Nitrospinae, Kiritimatiellaeota, "Candidatus Latescibacteria", "Candidatus Rokubacteria", Armatimonadetes, Elusimicrobia* and *Cyanobacteria.* The relative abundance of *Proteobacteria* varied between 25% and 55% depending on the treatment and season. It seems that sediment dredging increased the relative abundance of *Proteobacteria. Proteobacteria* phylum harbours a very diverse group of bacteria, some of which have been reported to responsible for N cycling in sediment (Fan et al., 2019), while others can reduce sulfate to hydrogen sulfide and thus promote the maintenance of anoxic conditions (Karnachuk et al., 2005). As has been reported, dredging may in some cases induce the release of nitrogen in sediment and consequently favor the growth of *Proteobacteria* (Yin et al., 2019). *In situ* inactivation by LMB did not notably change the relative abundance of *Proteobacteria* as compared to the control.

Representational difference analysis (RDA) was performed to visualize possible correlations between bacterial communities and P fractions in the sediments (Fig. 6b). The length of arrowheads show that both mobile P and HCl-P (Ca-P) can significantly ($p<0.05$) affect the bacterial communities in sediments and these were mainly driven by control treatment and LMB addition, respectively. Previous studies have reported that the input of P would be favor the growth of *Actinobacteria* in soil (Dai et al., 2020). For the first two sampling times the relative abundance of *Actinobacteria* was slightly higher in the control sediment than in LMB inactivated and dredged sediment, but in April 2019 the relative abundance also decreased in the control, indicating the influence of seasons in bacterial communities. Also, some study also indicated that the lack

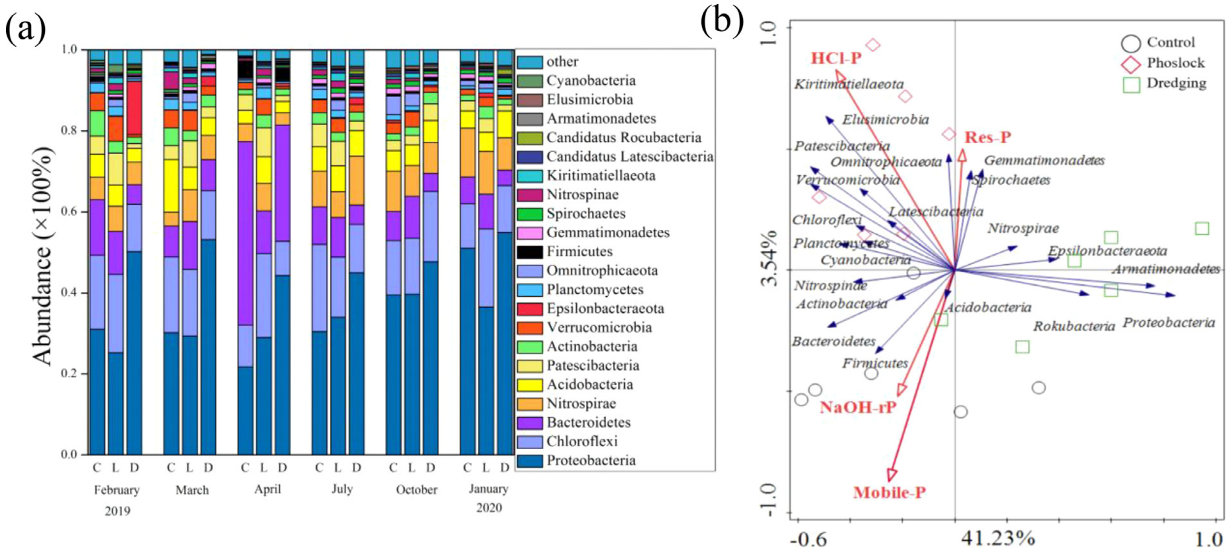


**Fig. 6.** Relative abundance of bacterial taxa at phylum level based on partial 16S rRNA gene sequencing of DNA extracted from the sediment in the control (C), LMB inactivated (L) and dredged (D) sediment treatments (a). Redundancy analysis (RDA) plot (b) of relationships between sediment P fractions and bacterial community composition.

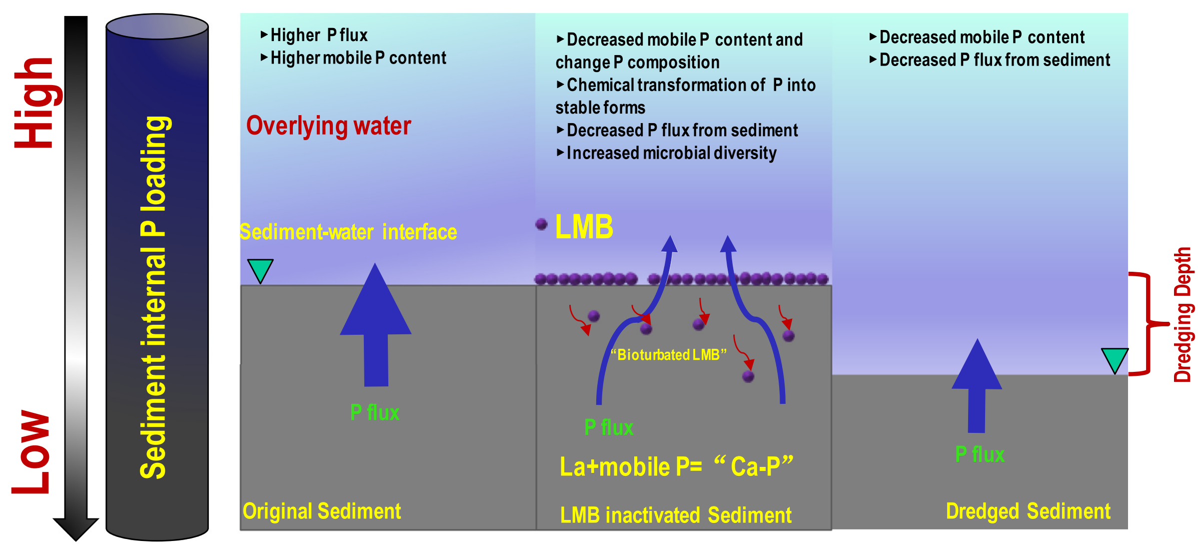


**Fig. 7.** Schematic representation of the internal phosphorus loading in original sediment (control treatment), *in situ* inactivation using LMB and dredged sediment.

of P would promote the growth of *Gemmatimonadetes* (Liang et al., 2020). The actual role of the detected bacteria in influencing the sediment P loading should be further investigated.

### 3.5. Mode of sediment internal p control by dredging and in situ inactivation and the implication for sediment management in eutrophic lakes

The aim of controlling internal sediment P loading in lakes is to persistently reduce the P fluxes across sediment-water interface over a substantial areal extent to reduce the mass (load) of labile P able to be utilised by algal biomass. Results of this and a previous study indicate that P fluxes are positively correlated with sediment mobile P (Fig. S2) (Ding et al., 2018a). Therefore, control of sediment mobile P can be considered an effective method to reduce internal P loading and the potential for P release in a eutrophic lake. Most commonly, this includes the two methods investigated here, that application of a P-binding/inactivation agent, in this case LMB, or the removal of surface sediment, often enriched in P via dredging.

In the present study, dredging and LMB inactivation have both been demonstrated to reduce the sediment mobile P concentration and consequently decrease P fluxes across sediment-water interface. A schematic representation of the respective mechanisms of dredging and *in situ* LMB inactivation on the sediment internal P loading is shown in Fig. 7. In sediment P inactivation an agent such as LMB can markedly increase sediment P adsorption capacity and decrease pore water P concentration. This is achieved by LMB transforming the sediment mobile P into a fraction that can be nominally equated with the Ca-P fraction. In contrast to the application of sediment P inactivation agents such as LMB, dredging aims to remove the uppermost organic- and nutrient-enriched sediment to reduce both the total nutrient inventory but also assimilable carbon in the organic matter that may facilitate the development of microbially-driven anoxic conditions leading to nutrient release. It is likely, however, that the effects of dredging gradually decrease over time due to the accumulation of new, often nutrient and organic matter-enriched sediment. In this scenario there is little prospect of P being converted into less mobile or even immobile forms with a greater propensity also for release during anoxic and or warmer period as demonstrated in this study. In contrast, the effects of the LMB are more pervasive in that bioturbation may periodically entrain and intimately mix both LMB and newly-deposited sediment into lower depths with labile P likely to be bound by LMB into less mobile forms, irrespective of temperature or redox status. However, future field large-scale experiment still should be carried out to investigate the two methods (dredging and *in situ* adsorbent inactivation) in the control of sediment internal P loading in eutrophic lakes.

## 4. Conclusion

A long-term field sediment cores incubation study was carried out to investigate the effectiveness of *in situ* inactivation using LMB and dredging for the control of sediment internal P loading in a eutrophic lake. Both LMB inactivation and dredging can decrease pore water P concentration and its flux from sediment when compared with control sediment. However, LMB addition may confer a more stable remediation option than sediment dredging for the control of pore water P concentration and fluxes as identified in this study during seasonal variation. Lanthanum-modified bentonite inactivation can significantly reduce the percentage of mobile P and increase the percentage of stable forms of P in relation to total P in sediment compared with the control sediment. In contrast, dredging did not change the percentage of mobile P relative to total P. Furthermore, LMB addition can increase the diversity of bacterial communities in sediment, whereas dredging appears to temporarily decrease bacterial diversity. The results of this study indicated that LMB inactivation may potentially offer improved management outcomes over dredging in the control of sediment internal P loading in eutrophic lakes. However, further research is required to investigate the long-term effects of the two methods and the role that microorganisms may play in P mobilization.

## Declaration of Competing Interest

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.

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## Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.watres.2020.116644.

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