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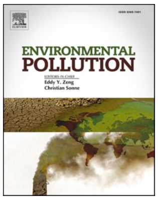

Response of spatio-temporal changes in sediment phosphorus fractions to vegetation restoration in the degraded river-lake ecotone☆

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ABSTRACT

Phosphorus (P) is an essential element in the ecosystem and the cause of the eutrophication of rivers and lakes. The river-lake ecotone is the ecological buffer zone between rivers and lakes, which can transfer energy and material between terrestrial and aquatic ecosystems. Vegetation restoration of degraded river-lake ecotone can improve the interception capacity of P pollution. However, the effects of different vegetation restoration types on sediment P cycling and its mechanism remain unclear. Therefore, we seasonally measured the P fractions and physicochemical properties of sediments from different restored vegetation (three native species and one invasive species). The results found that vegetation restoration significantly increased the sediment total P and bioavailable P content, which increased the sediment tolerance to P pollution in river-lake ecotone. In addition, the total P content in sediments was highest in summer and autumn, but lower in spring and winter. The total P and bioavailable P contents in surface sediments were the highest. They decreased with increasing depth, suggesting that sediment P assimilation by vegetation restoration and the resulting litter leads to redistribution of P in different seasons and sediment depths. Microbial biomass-P (MBP), total nitrogen (TN), and sediment organic matter (SOM) are the main factors affecting the change of sediment phosphorus fractions. All four plants' maximum biomass and P storage appeared in the autumn. Although the biomass and P storage of the invasive species *Alternanthera philoxeroides* were lower, the higher bioavailable P content and MBP values of the surface sediments indicated the utilization efficiency of sediment resources. These results suggest that vegetation restoration affects the distribution and circulation of P in river and lake ecosystems, which further enhances the ecological function of the river-lake ecotone and prevents the eutrophication and erosion of water and sediment in the river-lake ecotone.

1. Introduction

Phosphorus (P) is essential for biological growth and biogeochemical cycles (Vitousek et al., 2010). Therefore, it is generally considered a limiting factor in natural ecosystems (Hou et al., 2020). Under the action of river runoff, numerous point-source and diffused source phosphates have been partially input into the lake water, leading to different degrees of eutrophication of lakes and rivers (Schindler et al., 2016). The other part is input into the sediments of the river-lake ecotone, which has shrunk wetland areas and degenerated hygrophytes due to human activities. Artificial vegetation restoration in the river-lake ecotone is the most effective way to inhibit eutrophication and wetland degradation (Hilt et al., 2018; Li et al., 2021). Therefore, it is necessary to understand the effects of different vegetation restoration strategies on sediment P dynamics to develop more effective vegetation restoration strategies.

River-lake ecotone is the natural protective barrier of river and lake wetlands, which can carry out the exchange and transfer of capacity and material between terrestrial and aquatic ecosystems. It is an important area for P transport and deposition. It is also one of the primary sources,

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Abbreviations: P, phosphorus; MBP, microbial biomass phosphorus; AKP, alkaline phosphatase; BD, bulk density; MC, moisture content; SOM, sediment organic matter; EC, electrical conductivity; T, sediment temperature; TC, total carbon; TN, total nitrogen; TS, total sulfur; TP, total phosphorus; Pi, inorganic phosphorus; Po, organic phosphorus; dil. HCl-Pi, dilute HCl-Pi; conc. HCl-Pi, concentrated HCl-Pi; conc. HCl-Po, concentrated HCl-Po; Resid-P, Residual-P.

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sinks, and P converters in lakes and rivers (Johnson et al., 2014; Hongthanat et al., 2016). Part of the phosphorus transported by the external environment is absorbed into the biosphere by aquatic organisms, and more P is deposited in the sediment (Tang et al., 2020). P is present in organic and inorganic states in sediments, and sediment P dynamics is a complex biogeochemical process (Fu et al., 2020; Wang et al., 2021). In general, according to the degree of bioavailability, P fractions are divided into labile P, moderately labile P, and non-labile P (Rodrigues et al., 2016). Different P fractions generally reflect the effects of soil nutrient utilization and environmental factors (Maranguit et al., 2017; Wang et al., 2021). Labile P is often thought to be the P that plants can use during the growing season and is biologically available P (Rodrigues et al., 2016).

Vegetation restoration plays a crucial role in the dynamic changes of sediment P (Fu et al., 2020). Plants can alter soil abiotic factors, such as reducing soil pH and promoting the dissolution of numerous native mineral P, etc. (Jin et al., 2022). Part of the released fraction of inorganic P is absorbed and utilized by plants in the biosphere and then decomposed back into the sediment by falling matter (Dodd et al., 2018). At the same time, plants can also alter the dynamics of soil biological variables, such as changing sediment microbial community composition and sediment phosphatase activity, thereby affecting the mineralization and fixation of sediment P (Zhao et al., 2009; Fu et al., 2020). However, the transformation of sediment P (including the fixation or decomposition of different forms of P and adsorption or desorption) is also varied under different vegetation types (Grafe et al., 2018; Fan et al., 2021; Luo et al., 2021). For example, in the forest ecosystem, Fu et al. (2020) reported that the water-soluble inorganic P and organic P in the soil of the natural secondary forest after restoration were significantly higher than those of the *Eucalyptus* forest because the natural secondary forest had a higher litter decomposition rate than the *Eucalyptus* forest. In agroecosystems, Crews and Brookes (2014) reported that the unstable organic P pool in the soil of perennial grassland was significantly higher than that of annual wheat, and the inorganic P in the soil of annual wheat was significantly higher than in perennial grassland. It is possible that the microbial biomass-P (MBP) in perennial grassland is an order of magnitude greater than in annual wheat. These results suggest that the effects of plants on sediment P dynamics in different ecosystems depend on the plant type and growth properties. In addition, the river-lake ecotone is one of the more invasive severe places for alien species. Invasive species change the pattern of resource competition in the original communities, which inevitably affects the dynamics of sediment P (Wang et al., 2019). Therefore, understanding the dynamics of wetland vegetation and sediment P, as well as the ecological factors influencing river-lake ecotone, helps to promote vegetation restoration and sediment P utilization in wetland ecosystems and is crucial to the maintenance of ecosystem functions.

The dynamic changes of sediment P are also affected by seasonal changes in temperature and water composition, as well as rhythms in plant growth and development, which also lead to noticeable seasonal changes in sediment P dynamics (Norgbey et al., 2021). The seasonal dynamics of plant communities depend on changes in plant phenology (Mäkiranta et al., 2018). The microenvironment near the root system also varies due to plant growth and root activity across different growing seasons. Meanwhile, the quantity and quality of litter produced by plants in different seasons are also different. The rate of decomposition of this litter during different seasons and the rate of accumulation at the surface affect the return of nutrients such as nitrogen, P, and other substances to soil reservoirs (Mori et al., 2020). It is one of the critical links of the P cycle in the river-lake ecotone. Furthermore, the disturbance to sediment wet-dry alternation caused by water level fluctuations in different seasons can also affect the circulation of sediment P (Gao et al., 2020). However, in previous studies, less attention has been paid to the seasonal dynamics of sediment P in the river-lake ecotone and its drivers, which may hinder the maximization of the benefits of wetland ecological restoration.

Excessive human exploitation and reclamation at the base of Dongting Lake, the second-largest freshwater lake in China, have caused significant damage to the ecosystem for nearly 60 years. Among them, Lake Datong was separated from the mother lake, Dongting Lake, due to reclamation projects, and the vegetation in the river-lake ecotone was almost destroyed (Li et al., 2021). Since 2017, with the restoration of vegetation and the invasion of alien species in the river-lake ecotone of Lake Datong, it is ideal for us to study the dynamic changes of P fractions of sediments in different vegetation types and growth periods. Four major wetland vegetation types constructed from three native and one invasive species as the dominant species were selected for this study, respectively: native species *Phragmites australis*; *Zizania latifolia*; *Nelumbo nucifera*; and invasive species *Alternanthera philoxeroides*. At the same time, combined with different vegetation growth seasons and sediment depths, sediment P was fractioned to study its spatial and temporal variation. To this end, we have made the following assumptions:

(1) Vegetation restoration would change the physicochemical properties of sediments, which in turn would affect changes in sediment P fractions.
(2) The impact of different vegetation types on sediment P fractions is different, and exotic species could have a significantly different impact on P resources than native species.
(3) Biological processes were the main driving factor changes in P fractions in sediments.

2. Materials and methods

2.1. Study area and sample collection

Lake Datong (29°04′–29°22′ N, 112°17′–112°42′ E) is a sub-lake of Dongting Lake separated by dikes from the mother lake. It belongs to the continental monsoon humid climate zone. The annual average temperature is 16.5 °C, the annual average precipitation is 1240.8 mm, and the annual total solar radiation is 105.1 cal cm^{-2}. Among all the rivers flowing into and out of Lake Datong, the largest river-lake ecotone (sampled area) forms between the Jinpan Canal and Lake Datong (Fig. 1). The restoration of artificial ecological vegetation began in March 2017, and the primary vegetation needed to be restored were the native species of *P. australis*, *Z. latifolia*, and *N. nucifera*. During the recovery period, a small amount of *A. philoxeroides* grew in the recovery area and was manually removed during the recovery period. However, due to its strong reproductive capacity and the many asexual reproductive organs of *A. philoxeroides*, its remaining few reproductive organs rapidly formed stable vegetation. The phenological phase and growth habitats of these four plants are shown in supplementary Table S1. In order to explore the effects of different vegetation restoration types and different growth seasons on the P fractions of different sediment depths and to identify the main driving factor among these influencing factors. We selected the four dominant wetland vegetation types as the research objects for this study. Before vegetation restoration, we sampled the experimental field and measured the background value of sediment. There was no significant difference in sediment nutrient content among all the plots, with the nutrient contents of TN: 1.64 ± 0.16 mg g^{-1}, TC : 20.38 ± 1.11 mg g^{-1}, TP: 0.98 ± 0.08 mg g^{-1}, respectively. After artificial restoration in 2017, all the four dominant planting types were undisturbed. Detailed characteristics of the four plant community structures are shown in Table S2.

Five fixed sampling areas (10 × 10 m^2) were selected in each of the four restored plant communities and the mudflat bare land without vegetation cover. In order to minimize the impact of initial differences in the initial environmental conditions, the straight-line distance between the five sampling areas was less than 500 m. In 2020, we collected plant and sediment samples for the four growth seasons of plants, including the seeding period (spring, April), the growing period (summer, June),

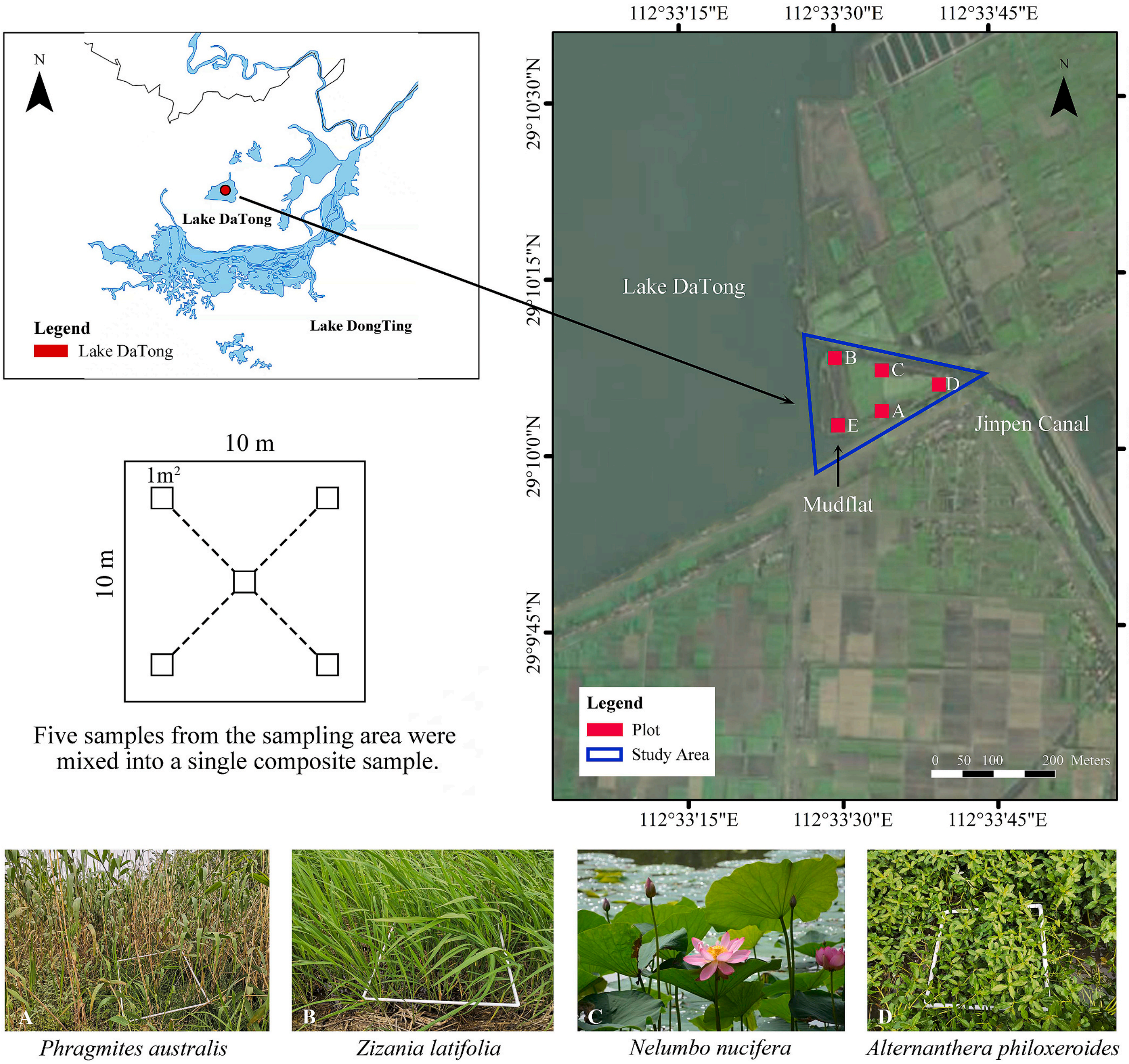


Fig. 1. Long-term sampling sites in the river-lake ecotone formed by Lake Datong and Jinpen Canal in the Dongting Lake Basin in China. The capital letters A, B, C, D in the figure represent the locations of the four plant communities. A is the *Phragmites australis*, B is the *Zizania latifolia*, C is the *Nelumbo nucifera*, D is the *Alternanthera philoxeroides* and E is the mudflat bare land.

the flowering and fruiting period (autumn, September), and the decay period (winter, December). Above all, a quadrangle (1 × 1 m^2) was set in the sampling division to collect the plants' biomass. Then, sediment mud columns with a depth of 0–40 cm and a diameter of 5.95 cm were collected from five randomly selected points using a soil column sampler (Corer, Uwitec, Austria). Five samples from each sampling area were mixed and homogenized to form a single composite sample. The fresh sediment samples were screened through 18 mesh holes (1 mm pore) to remove impurities such as small stones, plant roots and shells of benthic animals. After screening, sediment samples were divided into two parts. One part was stored at 4 °C for measuring microbial biomass-P (MBP) and alkaline phosphatase activities (AKP). Another part was used to measure the physicochemical indicators and the P fractions of the sediment after drying in the shade.

2.2. *Determination of the physicochemical properties of the sediment*

Separately, another unmixed sediment column was taken to measure the sediment bulk density (BD). The 0–10 cm soil layer was measured using a column of sediment samples taken from the sampling tube (Zhou et al., 2016). The volume of removed sediment could be measured by backfilling it with a known volume of water, and the remaining three layers of sediment BD could be measured by the cutting ring method. Sediment moisture content (MC) was measured by drying at 75 °C for 48 h. Sediment organic matter (SOM) content was measured by the loss of sediment samples after 6 h of combustion at 550 °C in a muffle furnace (KSL-1200X, HF-Kejing, China). A pH meter (Hanna Instruments HI99121, Italy) was used to measure sediment pH, and a WET sensor (Delta-T Devices, Cambridge, UK) was used to determine electrical conductivity (EC) and temperature (T). The contents of sediment total carbon (TC), total nitrogen (TN) and total sulfur (TS) were measured by an elemental analyzer (Elementar UNICUBE, Germany).

2.3. Determination of P fractions and total P in plants and sediments

The sediment phosphorus fractions are determined by the modified Hedley Fractionation Technique (Tiessen and Moir, 1993). The specific process is shown in Fig. S1. The method is to extract sediment P in sequence from weak to strong extractants, which are anion resin exchange membrane (0.9 × 0.62 cm, bicarbonate form), 0.5 M $NaHCO_3$, 0.1 M NaOH, 1 M HCl, hot concentrated HCl and concentrated H_2SO_4 with 30% H_2O_2. Weighed 0.5 g of dry soil through a 100-mesh sieve. After the extraction agent was used to extract and adjust the pH, the concentration of inorganic P (Pi) in each extract was determined by the molybdate-ascorbic acid method, and the concentration of total P (TP) in the extract was determined by persulfate digestion. The difference between TP and Pi is the organic P (Po) content. Nine different sediment P fractions can be obtained by sequence extraction, which are Resin-Pi, $NaHCO_3$-Pi, $NaHCO_3$-Po, NaOH-Pi, NaOH-Po, dilute HCl-Pi (dil. HCl-Pi), concentrated HCl-Pi (conc. HCl-Pi), concentrated HCl-Po (conc. HCl-Po) and Residual-P (Resid-P). The roles of these nine P fractions in biogeochemical cycles are shown in Table S3. In addition, the plants were washed with deionized water and dried to a constant mass (75 °C, 48–72 h). After weighing, the plants were digested with sulfuric acid and 30% hydrogen peroxide. The concentration of P was determined by the molybdate/ascorbic acid method (Yan et al., 2021). Sediment total P was measured by the Standard Measurements Testing Program of the European Union (SMT) (Ashraf P et al., 2006). Each of the above indexes was repeated three times.

2.4. Determination of sediment AKP and MBP assay

The sediment AKP activity is determined by the p-nitrophenyl phosphate disodium method. We took 1 g of fresh sediment sample, used p-nitrophenyl phosphate (pNPP) as the substrate, performed the enzymatic reaction at 37 °C, pH = 11 buffer, stopped the reaction with 3 mol NaOH after 1 h, and centrifuged at 4000 rpm for 20 min. Then we took the absorbance of the supernatant at 410 nm to determine the sediment AKP activity and expressed it as mg p-nitrophenol kg^{-1} dry soil h^{-1}.

Sediment MBP was determined by chloroform fumigation and $NaHCO_3$ extraction of inorganic phosphorus (Chen et al., 2016). Three fresh 10 g sediment samples were taken, and the first part was added to 0.5 mol $NaHCO_3$ at a soil to water ratio of 1:20 (W: V), shaken for 1 h (200 rpm), and filtered to determine the P content. The second part was added to chloroform and fumigated at 25 °C in the dark for 24 h and then followed the same procedure as the first part. The third part was extracted by adding 0.5 mol KH_2PO_4 and 0.5 mol $NaHCO_3$ to determine the recovery rate of the added orthophosphate inorganic P to correct the absorption and fixation of the MBP released by the fumigation treatment of the sediment. The MBP is quantified using the following equation:

$$\mathrm{MBP} = E_{pi} / (K_p \times R_{pi})$$

where the E_{pi} is the difference between the second fumigation part and the first non-fumigation part. R_{pi} is the difference between the third addition to the orthophosphate part and the first non-fumigation part. K_p is the conversion factor, which is 0.40 (Jenkinson, 2004).

2.5. Statistical analysis

We used a three-way analysis of variance (ANOVA) to test the effects of growth season (GS), vegetation restoration type (VR), and sediment depth (SD) on the sediment P fractions dynamic in the river-lake ecotone. The differences in sediment physicochemical properties of different restoration types and sediment depths were tested by Duncan's test for multiple comparisons. Before the data analysis, a $\log_{10}(x)$ transformation was performed to satisfy the normal distribution. Differences were considered significant at $P < 0.05$. This part of the data analysis was carried out using SPSS 23.0 (SPSS, Chicago, Illinois, USA). We then used the "vegan" package for a redundancy analysis (RDA) to study the potential sediment physicochemical properties that affect the dynamic changes of sediment P fractions. Meanwhile, a hierarchical analysis was performed using the "rdacca. hp" package to obtain the contribution of the physicochemical properties of individual sediments to the changes in sediment P fractions (Lai et al., 2022). Pearson correlation analysis was used to study the relationship between sediment P fractions and sediment properties. All regressions and correlations were considered significant at $P < 0.05$. Data analysis in this section was performed using R software (version 4.0.3).

3. Results

3.1. Seasonal variations in vegetation biomass and P stocks

The temporal dynamics of the aboveground and belowground biomass of the four plants during the growing season are shown in Fig. 2A, with a typical single-peak curve. Plants grow rapidly from June, and the biomass reaches its maximum in September. Among them, *P. australis* had the largest aboveground biomass, attaining 3862.75 g m^{-2}, and *N. nucifera* had the largest belowground biomass, reaching 387.37 g m^{-2} (Fig. 2A). The P storage in planting objects showed apparent seasonal variation ($P < 0.05$) (Fig. 2B). The minimum value of aboveground P storage appeared in April, and the minimum value of belowground P storage appeared in December. The maximum value of aboveground and belowground reserves both occurred in September. There were also significant differences in P storage among plants ($P < 0.05$) (Fig. 2B). The pattern of P storage was the same as that of plant biomass, and the aboveground part of *P. australis* had the maximal P storage, reaching 19.55 mg per unit area, and the belowground part of *N. nucifera* had the maximal P storage, reaching 1.44 mg per unit area (Fig. 2B).

3.2. Sediment physicochemical properties

The vegetation restoration types had significant effects on the physicochemical properties of the sediment ($P < 0.05$, Table 1). Compared with bare land in mudflats, vegetation restoration significantly increased the content of sediment MC, SOM, TN, TC, and TP. Among all vegetation restoration types, the nutrient TN, TC, and TP content of sediment restored by *N. nucifera* were the highest, and the pH was the lowest ($P < 0.05$, Table 1). In addition, there were significant differences in the physicochemical properties of different depths of sediments across all vegetation restoration types. The contents of nutrients (TN, TC, and TP), MC, and SOM in the 0–10 cm surface layer were the highest, which decreased significantly with the increase of depth, while pH, EC, and BD increased significantly with the increase of depth ($P < 0.05$, Table 1).

3.3. Sediment P dynamics

The results of a three-way analysis of variance showed that, except for conc. HCl-Pi/Po, the three factors, including vegetation restoration types, growing seasons, and sediment depth, and the interaction of the three factors had significant effects on the remaining P fractions ($P < 0.05$, Table 2). The maximum and minimum values of the total P of the sediments appeared in the growing period (June) and decay period (December) of plants, respectively, as shown in Fig. 3. The sum of P fractions in sediments near *N. nucifera* roots was the highest at the seeding period and the flowering and fruiting period, and in sediments near *A. philoxeroides* roots was the highest at the growing period and decay period. While the sum of P fractions in the sediment of mudflat bare land was the lowest throughout the year. With the increase of

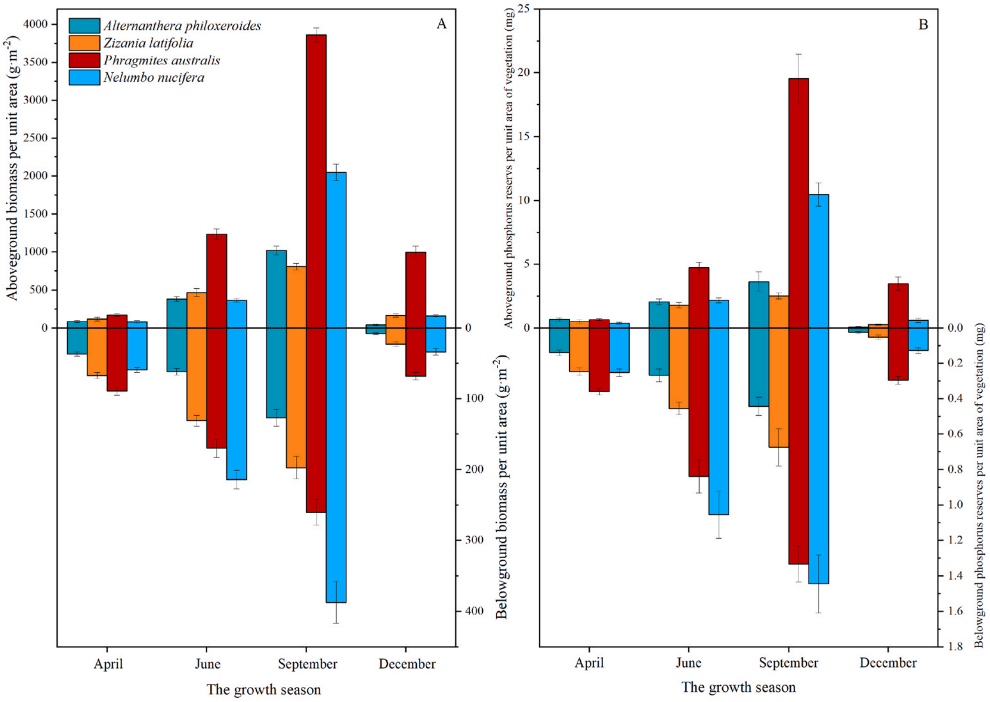


Fig. 2. Seasonal dynamics of aboveground and belowground biomass and phosphorus storage per unit area of different vegetation in the river-lake ecotone. (A) shows the aboveground and belowground biomass, and (B) shows the phosphorus storage per unit area.

sediment depth, the total P fractions of the sediment were shown to decline, and the values of 0–10 cm in the surface layer were relatively the largest (Fig. 3). Among all the P fractions, Resin-Pi had the largest difference between surface and lower layers, with a content of 0–10 cm in the surface layer of 29.06 mg kg^{-1}-165.12 mg kg^{-1}, and the content in the lower layer of 30–40 cm was only 10.46 mg kg^{-1}-86.26 mg kg^{-1}. The contents of labile P and moderately labile P and non-labile P in all samples were 26.24 mg kg^{-1}-426.15 mg kg^{-1} and 386.32 mg kg^{-1}-1012.20 mg kg^{-1} and 210.12 mg kg^{-1}-345.93 mg kg^{-1}, respectively.

Among all the P fractions, the total amount of inorganic P accounted for 85.58–95.72% of the total P fractions, and the total amount of organic P accounted for only 4.28–14.42% of the total P fractions (Fig. 4). The proportions of Resin-Pi, $NaHCO_3$-Pi, $NaHCO_3$-Po, NaOH-Pi, NaOH-Po, dil. HCl-Pi, conc. HCl-Pi, conc. HCl-Po and Resid-P in the total P fractions were 1.08–10.58%, 2.06–17.01%, 0.22–2.92%, 3.86–31.17%, 0.49–10.90%, 26.81–57.11%, 7.13–15.95%, 2.50–9.87% and 5.67–20.03%, respectively. Furthermore, the proportion of labile P fraction of Resin-Pi and $NaHCO_3$-Pi/Po was significantly higher than that of mudflat bare land in the unrestored area, especially in 0–10 cm surface sediments. On the contrary, the P fractions of dil. HCl-Pi, conc. HCl-Pi/Po and Resid P, the proportion of non-labile P in the mudflat bare land were significantly higher than that in the sediment of vegetation restoration. With the increase in sediment depth, the proportion of labile P in all sediments tended to decrease, while the proportion of non-labile P increased. Moreover, the proportion of labile P in the sediment was the largest in the growing period and the smallest in the decay period. The proportion of non-labile P was reversed (Fig. 4).

3.4. MBP content and AKP activity

Both the MBP content and AKP activity varied significantly between vegetation types and growth seasons. Under *A. philoxeroides* vegetation, MBP content and AKP activity were significantly higher than those of other vegetation types during most of the growth season, while MBP content and AKP activity were lower than those of other vegetation types on mudflat bare land almost all year round (Fig. 5).

Sediment MC and temperature had a significant positive correlation with sediment AKP activity, with correlation coefficients of $r = 0.57$ and $r = 0.45$, respectively. Except for that pH, EC and BD had a significant negative correlation with MBP, the other sediment physicochemical properties all had a significant positive correlation with sediment MBP, and there was also a significant positive correlation between AKP activity and MBP, with a correlation coefficient of $r = 0.47$ (Fig. 6).

3.5. Correlations related sediment properties to P fractions

The results of the redundancy analysis showed that the first two axes of the redundancy analysis accounted for 68.10% of the dynamic changes of the sediment P fractions ($P < 0.001$) (Fig. 7A). For individual sediment physicochemical properties, MBP ($R^2 = 0.12$) contributed the most degree of variance of the all the physicochemical properties, followed by TN ($R^2 = 0.11$), SOM ($R^2 = 0.09$), TS ($R^2 = 0.07$), pH ($R^2 = 0.07$), MC ($R^2 = 0.07$), BD ($R^2 = 0.06$), TC ($R^2 = 0.06$), AKP ($R^2 = 0.02$), and EC ($R^2 = 0.01$) (Fig. 7B). Except for the three sediment physicochemical properties, pH, BD and EC, which were negatively correlated with sediment P fractions, all other sediment physicochemical properties were positively correlated with the sediment P fractions. MBP and MC showed a significant positive correlation with all sediment P fractions ($P < 0.05$) (Fig. 6).

4. Discussion

4.1. Effects of vegetation restoration on the physicochemical properties of the sediment

Vegetation restoration has received much attention, as an essential measure to reduce soil erosion and pollutant interception in the river-lake ecotone (Cerdà et al., 2021; Xiao et al., 2021). In addition to preventing soil erosion and reducing pollution runoff, vegetation restoration can alter sediment soil ecosystem processes (Zhang et al., 2019). Studies have shown that vegetation restoration can reduce soil erosion and improve the quality of sediment-soils (Albaladejo et al., 1998; Su and Zhao, 2003; Gao et al., 2014). Meanwhile, vegetation restoration improves soil aggregate structure, nutrient status, sediment pH and

Table 1
Comparison of sediment physicochemical properties of vegetation restoration types and sediment depths during the plant growing season.

Vegetation type	Depth	Sediment physicochemical properties							
		pH	MC	EC (ms·m^{-1})	SOM (mg·g^{-1})	BD (g·cm^{-3})	TN (mg·kg^{-1})	TC (mg·kg^{-1})	TP (mg·kg^{-1})
A. philoxeroides	0–10	8.13 ± 0.02Ca	0.63 ± 0.01Cc	95.75 ± 1.16Ba	106.22 ± 0.04BCc	0.73 ± 0.01Ba	2337.00 ± 14.43Ba	25396.25 ± 28.87Ba	1480.47 ± 69.52Ca
	10–20	8.25 ± 0.01Cb	0.52 ± 0.00Cb	116.31 ± 0.76Bb	78.01 ± 2.78BCa	0.75 ± 0.01Bb	1927.25 ± 20.21Bb	23307.00 ± 58.31Bb	1085.70 ± 57.41Cb
	20–30	8.49 ± 0.01Cc	0.42 ± 0.01Ca	118.63 ± 1.02Bc	84.79 ± 0.16BCb	0.82 ± 0.02Bc	1710.50 ± 10.97Bc	21237.25 ± 51.38Bc	860.57 ± 39.76Cc
	30–40	8.61 ± 0.01Cd	0.41 ± 0.00Ca	119.06 ± 1.61Bd	83.88 ± 0.22BCb	0.88 ± 0.01Bd	1592.75 ± 16.17Bd	19848.50 ± 69.28Bd	809.25 ± 36.39Cc
Z. latifolia	0–10	8.12 ± 0.02ABa	0.60 ± 0.00Dc	92.44 ± 1.06Da	114.32 ± 0.07Cd	0.73 ± 0.01Ca	2513.00 ± 20.21Da	26627.25 ± 50.81Ca	1303.44 ± 73.38Ca
	10–20	8.22 ± 0.02ABb	0.49 ± 0.00Db	124.69 ± 0.62Db	100.46 ± 0.54Cc	0.82 ± 0.02Cb	2141.25 ± 16.17Db	24019.50 ± 57.16Cb	958.70 ± 50.32Cb
	20–30	8.33 ± 0.01ABc	0.48 ± 0.00Da	131.06 ± 0.97Dc	81.28 ± 0.15Ca	0.87 ± 0.01Cc	1732.25 ± 14.43Dc	22516.00 ± 63.51Cc	874.74 ± 45.91Cc
	30–40	8.47 ± 0.01ABd	0.47 ± 0.00Da	133.88 ± 0.04Dd	83.65 ± 0.23Cb	0.94 ± 0.01Cd	1746.00 ± 19.05Dd	22743.75 ± 43.30Cd	898.00 ± 47.13Cc
P. australis	0–10	8.22 ± 0.01Ba	0.53 ± 0.01Bc	104.06 ± 0.87Ea	91.50 ± 0.04Bd	0.80 ± 0.01Da	2164.18 ± 18.48Ca	25996.14 ± 39.26Da	987.74 ± 55.80Ba
	10–20	8.23 ± 0.01Bb	0.45 ± 0.01Bb	130.02 ± 0.65 Eb	85.96 ± 1.10Bc	0.86 ± 0.01Db	1865.50 ± 12.70Cb	24544.25 ± 45.03Db	879.07 ± 52.65Bb
	20–30	8.32 ± 0.01Bc	0.44 ± 0.01Ba	130.75 ± 0.63Ec	80.78 ± 0.09Ba	0.90 ± 0.01Dc	1836.25 ± 14.43Cc	24312.75 ± 63.51Dc	856.77 ± 48.90Bc
	30–40	8.42 ± 0.01Bd	0.42 ± 0.01Ba	132.94 ± 0.92Ed	83.11 ± 0.41Bb	0.98 ± 0.01Dd	1852.25 ± 19.63Cd	24279.00 ± 37.53Dd	853.51 ± 48.37Bc
N. nucifera	0–10	8.10 ± 0.01Aa	0.63 ± 0.01Ec	95.38 ± 1.19Ca	97.44 ± 0.49BCa	0.67 ± 0.02Aa	2638.75 ± 14.43Ea	26125.50 ± 69.28Ea	1444.06 ± 50.67Da
	10–20	8.24 ± 0.01Ab	0.57 ± 0.01 Eb	112.81 ± 2.12Cb	84.82 ± 7.81BCa	0.72 ± 0.01Ab	2446.50 ± 10.39 Eb	25494.50 ± 46.19 Eb	1280.94 ± 58.66Db
	20–30	8.30 ± 0.03Ac	0.54 ± 0.01Ea	123.13 ± 1.71Cc	88.78 ± 1.22BCa	0.78 ± 0.01Ac	2139.00 ± 12.70Ec	24534.00 ± 32.91Ec	1115.56 ± 43.55Dc
	30–40	8.41 ± 0.02Ad	0.51 ± 0.02Ea	126.63 ± 1.69Cd	84.81 ± 0.07BCa	0.86 ± 0.02Ad	1942.00 ± 8.50Ed	23920.00 ± 23.09Ed	847.36 ± 44.47Dc
Mudflat	0–10	8.18 ± 0.03Ca	0.51 ± 0.01Ac	83.94 ± 1.59Aa	76.24 ± 0.65Ad	0.82 ± 0.01Ea	1589.50 ± 11.55Aa	22378.50 ± 46.19Aa	912.02 ± 47.87Aa
	10–20	8.26 ± 0.01Cb	0.43 ± 0.01Ab	100.13 ± 1.68Ab	70.74 ± 0.48Ac	0.87 ± 0.02 Eb	1434.25 ± 9.24Ab	21172.25 ± 63.51Ab	709.31 ± 37.23Ab
	20–30	8.44 ± 0.02Cc	0.39 ± 0.01Aa	116.69 ± 1.84Ac	59.81 ± 0.05Aa	0.94 ± 0.02Ec	1171.75 ± 15.01Ac	19425.50 ± 51.96Ac	603.28 ± 31.66Ac
	30–40	8.53 ± 0.01Cd	0.43 ± 0.01Aa	119.94 ± 1.37Ad	61.99 ± 0.34Ab	1.00 ± 0.01Ed	1001.25 ± 12.70Ad	18240.25 ± 45.03Ad	625.09 ± 32.81Ac

Note: pH, sediment pH; MC, sediment moisture content; EC, electrical conductivity; SOM, sediment organic matter; BD, bulk density; TN, total nitrogen; TC, total carbon; TP, total phosphorus.
Data indicates Means ± Standard error (SE).
Different capital letters indicate significant measured values within among different vegetation types in the same depth and different lowercase letters indicate significant measured values among in different sediment depth in same vegetation type (Duncan's test, $P < 0.05$).

Table 2
Effects of growth seasons (GS), vegetation restoration types (VR) and sediment depth (SD) on the sediment phosphorus fractions in the river-lake ecotone using three -way ANOVA (values in bold are below the significance level of 0.05).

	GS		VR		SD		GS × VR		GS × SD		VR × SD		GS × VR × SD	
	F *(3,160)*	*P*	*F* *(4,160)*	*P*	*F* *(3,160)*	*P*	*F* *(12,160)*	*P*	*F* *(9,160)*	*P*	*F* *(12,160)*	*P*	*F* *(36,160)*	*P*
Resin-Pi	2158.87	**<0.001**	1331.92	**<0.001**	2067.80	**<0.001**	398.74	**<0.001**	129.72	**<0.001**	182.79	**<0.001**	50.06	**<0.001**
NaHCO$_3$-Pi	342.7	**<0.001**	594.50	**<0.001**	964.97	**<0.001**	164.32	**<0.001**	42.29	**<0.001**	63.69	**<0.001**	19.52	**<0.001**
NaHCO$_3$-Po	20.22	**<0.001**	11.69	**<0.001**	6.30	**<0.001**	4.92	**<0.001**	3.28	**0.001**	2.37	**0.008**	3.08	**<0.001**
NaOH-Pi	607.35	**<0.001**	1957.87	**<0.001**	2652.92	**<0.001**	602.47	**<0.001**	81.89	**<0.001**	183.89	**<0.001**	69.22	**<0.001**
NaOH-Po	9.19	**<0.001**	6.99	**<0.001**	57.46	**<0.001**	15.25	**<0.001**	5.15	**<0.001**	8.01	**<0.001**	4.80	**<0.001**
dil. HCl-Pi	49.49	**<0.001**	44.43	**<0.001**	111.31	**<0.001**	38.10	**<0.001**	14.5	**<0.001**	17.91	**<0.001**	9.48	**<0.001**
conc. HCl-Pi	9.99	**<0.001**	37.86	**<0.001**	26.89	**<0.001**	2.64	**0.003**	2.49	**0.011**	2.57	**0.004**	0.68	0.910
conc. HCl-Po	2.63	0.052	12.64	**<0.001**	3.64	**0.014**	2.71	**0.002**	0.68	0.731	4.11	**<0.001**	0.90	0.639
Resid-P	12.89	**<0.001**	78.87	**<0.001**	155.30	**<0.001**	20.85	**<0.001**	3.99	**<0.001**	15.19	**<0.001**	4.64	**<0.001**

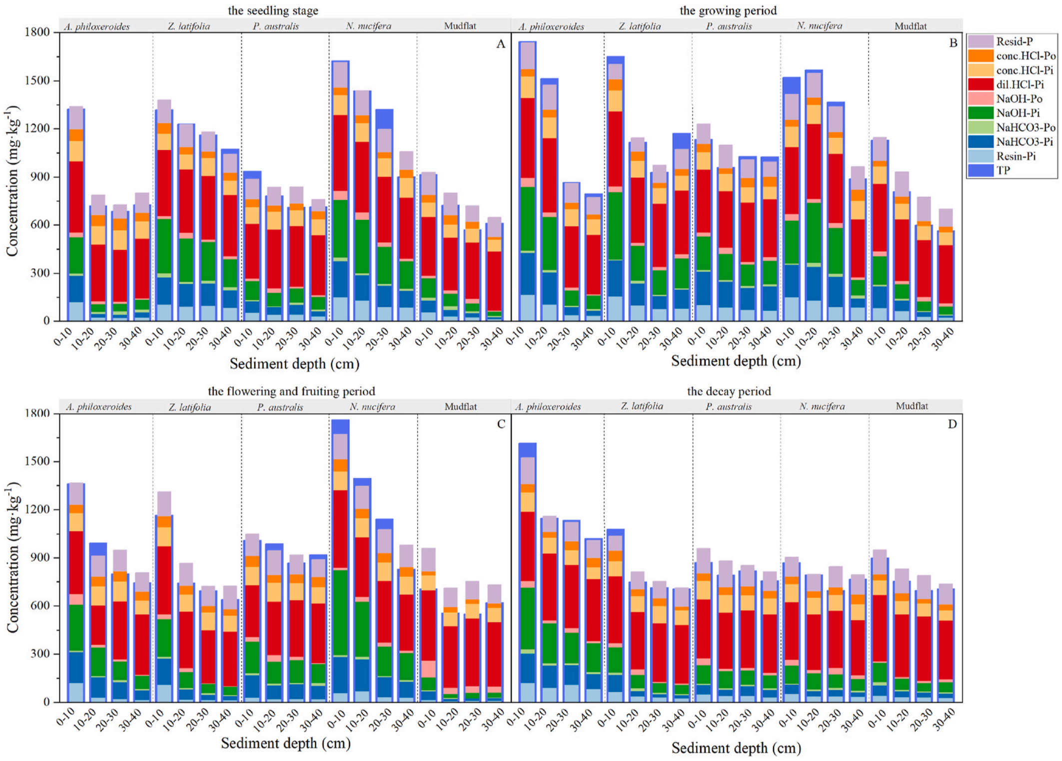


Fig. 3. Dynamic of sediment phosphorus fractions in river-lake ecotone at different growth season, vegetation restoration type and sediment depth.

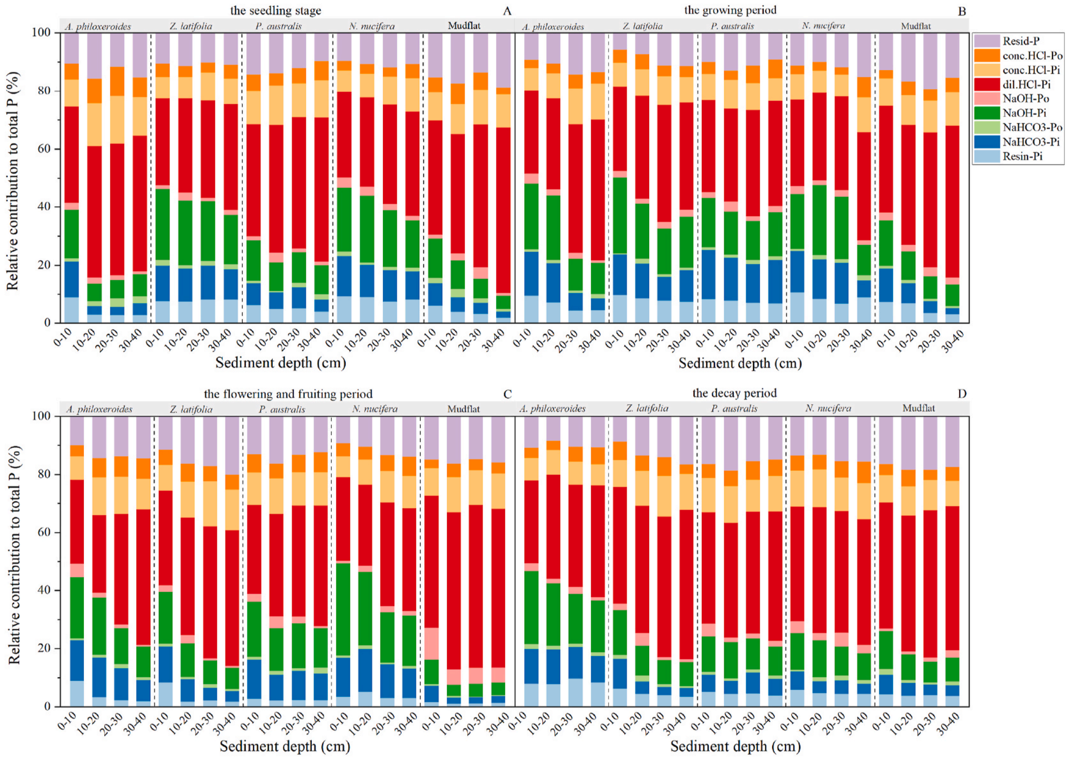


Fig. 4. The percentage of sediment phosphorus fractions relative to total phosphorus fractions under different growth season, vegetation restoration type and sediment depth in the river-lake ecotone.

hydrothermal conditions. As well as organic matter, sediment microbial and enzyme activities increased (Zhang et al., 2019; Ovsepyan et al., 2020). In this study, there were significant differences in the dynamics of pH, MC, EC, SOM, BD, TN, TC and TP content of sediment in the unrestored mudflat land and sites of different vegetation restoration types. The soil fertility of sediment was significantly higher after vegetation restoration than in the mudflat bare land. The content of SOM was also the lowest in mudflat bare land. This is because the accumulation

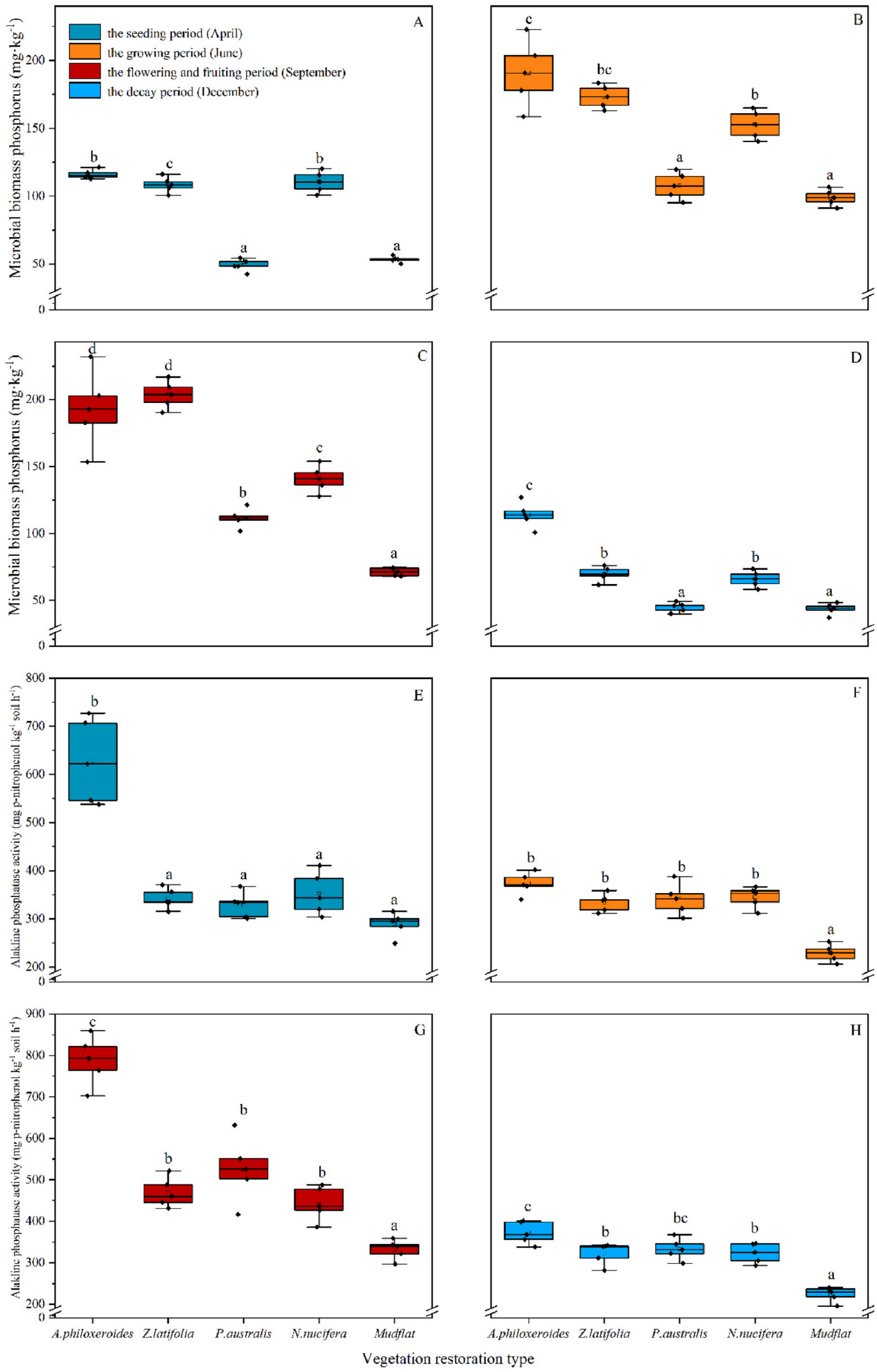
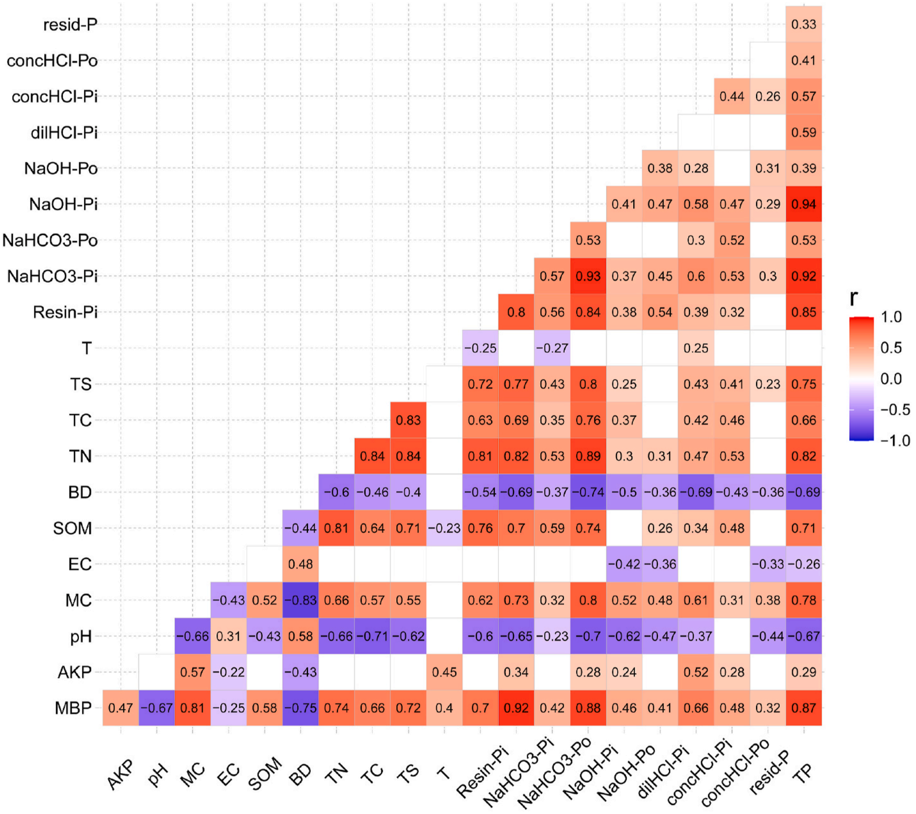


Fig. 5. Boxplots of microbial biomass phosphorus content and alkaline phosphatase activity in surface 0–10 cm sediments for each growing season and vegetation restoration type. Lower and upper box boundaries represent the quartiles (25% and 75% quantiles, respectively), the whisker is min––max, the black small square is the mean value, and the solid lines across each box are the median. Different lowercase letters indicate significant differences ($P < 0.05$) among vegetation types using Duncan's test.

and mineralization of dead leaves and roots decomposing materials generated by plant growth in sediments enhance the organic matter in sediments, and the sediments TN, TC and TP are strongly correlated with organic matter. After all, they are mainly derived from organic matter in sediments (Cleveland and Liptzin, 2007). The physicochemical properties of sediments also differ significantly among different vegetation types, indicating the heterogeneity in the impact of different vegetation types on sediments (Fu et al., 2020). These differences can help clarify the impact of vegetation ecological restoration on the sediment development path of the river-lake ecotone.

4.2. Effects of vegetation restoration on P content and fractions of sediment

In this study, the total P content in sediments increased significantly after vegetation restoration compared with unrestored mudflat bare



	AKP	pH	MC	EC	SOM	BD	TN	TC	TS	T	Resin-Pi	NaHCO3-Pi	NaHCO3-Po	NaOH-Pi	NaOH-Po	dilHCl-Pi	concHCl-Pi	concHCl-Po	resid-P	TP
resid-P																				0.33
concHCl-Po																				0.41
concHCl-Pi																		0.44	0.26	0.57
dilHCl-Pi																				0.59
NaOH-Po																0.38	0.28		0.31	0.39
NaOH-Pi															0.41	0.47	0.58	0.47	0.29	0.94
NaHCO3-Po														0.53			0.3	0.52		0.53
NaHCO3-Pi													0.57	0.93	0.37	0.45	0.6	0.53	0.3	0.92
Resin-Pi												0.8	0.56	0.84	0.38	0.54	0.39	0.32		0.85
T											−0.25		−0.27				0.25			
TS											0.72	0.77	0.43	0.8	0.25		0.43	0.41	0.23	0.75
TC									0.83		0.63	0.69	0.35	0.76	0.37		0.42	0.46		0.66
TN								0.84	0.84		0.81	0.82	0.53	0.89	0.3	0.31	0.47	0.53		0.82
BD							−0.6	−0.46	−0.4		−0.54	−0.69	−0.37	−0.74	−0.5	−0.36	−0.69	−0.43	−0.36	−0.69
SOM						−0.44	0.81	0.64	0.71	−0.23	0.76	0.7	0.59	0.74		0.26	0.34	0.48		0.71
EC						0.48									−0.42	−0.36			−0.33	−0.26
MC				−0.43	0.52	−0.83	0.66	0.57	0.55		0.62	0.73	0.32	0.8	0.52	0.48	0.61	0.31	0.38	0.78
pH			−0.66	0.31	−0.43	0.58	−0.66	−0.71	−0.62		−0.6	−0.65	−0.23	−0.7	−0.62	−0.47	−0.37		−0.44	−0.67
AKP			0.57	−0.22		−0.43				0.45		0.34		0.28	0.24		0.52	0.28		0.29
MBP	0.47	−0.67	0.81	−0.25	0.58	−0.75	0.74	0.66	0.72	0.4	0.7	0.92	0.42	0.88	0.46	0.41	0.66	0.48	0.32	0.87

Fig. 6. The matrix plot of Pearson correlation coefficients for soil properties and phosphorus fractions. The stronger the color, the greater the correlation coefficient, and only significant correlations ($P < 0.05$) were included. The color and number represent the Pearson's correlation coefficient (blue ~ red : -1–1). (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

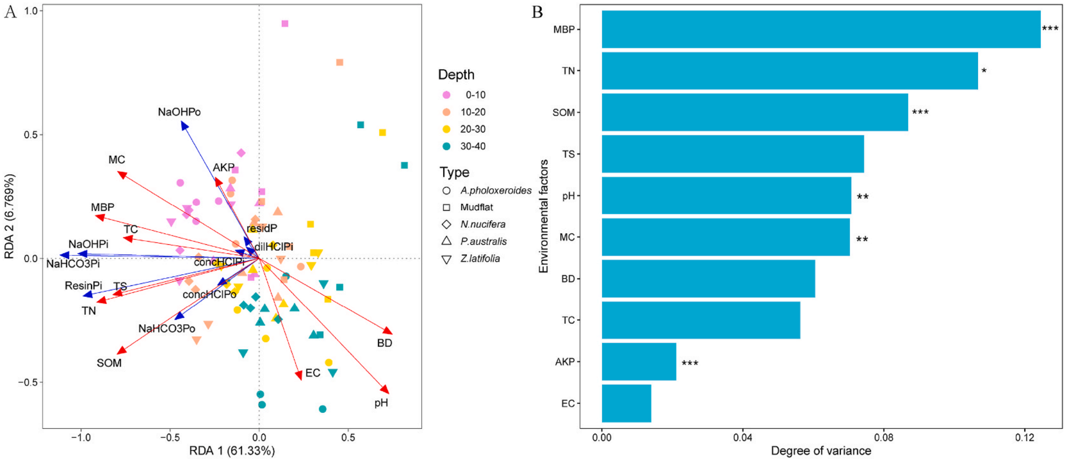


Fig. 7. Redundancy analysis showed the effect of sediment properties on sediment phosphorus fractions. (A) shows the RDA plot of the phosphorus fractions, sediment physicochemical properties and samples. The explained variations of the first and second axes are given in brackets. Vectors represent nine phosphorus fractions with blue arrows; the red arrows indicate the physicochemical properties of sediment; samples are represented by solid circles of different shapes. (B) shows the effects of sediment physicochemical properties on the content of phosphorus fractions. Significance levels are indicated by asterisks: ***$p < 0.001$, **$p < 0.01$, *$p < 0.05$. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

land. The proportion of bioavailable labile P and moderately labile P increased significantly in the sediments. These results indicate that vegetation restoration can effectively increase the P content in the sediments so that the P fractions can gradually migrate in a direction more beneficial to plant growth and absorption. The impact of this change is significant. Due to the unique geographical location of the river-lake ecotone, environmental changes and human disturbances often occur. Geochemical behaviors such as material migration and nutrient conversion occur between the interface of sediment and water, and sediment acts as the "source and sink" of material (Krishna Prasad and Ramanathan, 2008), and the content of P stored in sediments is also constantly changing. Vegetation restoration increases the P storage capacity of sediments in the river-lake ecotone. It better plays to the ecological function of adsorption, settlement, and interception of nutrients in the ecotone, thus further reducing the nutrient input from land to water and ensuring the safety of lake water.

At the same time, the study also found that the proportion of dil. HCl-Pi content in the sediment was lower than in the unrestored mudflat bare land after vegetation restoration. As a non-labile P, the dil. HCl-Pi is not easily absorbed and utilized directly by plants, but the content of dil. HCl-Pi is consumed by the restoration of vegetation. Previous studies have shown that HCl–P concentrations are deficient in acidic environmental sediments (Fu et al., 2020). This study found that vegetation restoration could control the variation of sediment pH. Plant root secretions, litter decomposition and microbial activities reduce the pH of sediment (Vincent et al., 2011; Jin et al., 2022), and the acidic environment caused by the pH reduction dissolves the apatite, which is less resistant to weathering, resulting in the release of a large amount of dil. HCl-Pi from the sediment (Li and Brett, 2013). The released dil. HCl-Pi has three directions in the wetland sediments of the river-lake ecotone. One part enters the lake water through the interstitial water; the other part is absorbed by metal oxides such as iron, aluminum, and manganese in the sediment and becomes metal-bound P, but this part of P is challenging for plants to absorb and utilize. Another part is absorbed and utilized by plants and microorganisms to become biomass P (Pang et al., 2009). It returns to the sediments in the form of organic P through litter decomposition after the death of the organisms (Zhu et al., 2021). These also explain the phenomenon mentioned above that the proportion of labile P increases after vegetation restoration.

There were also significant differences in sediment P fractions under different vegetation types, and the differences are mainly reflected in the labile and moderately labile P in sediments. In general, the difference in sediment P fractions caused by such differences between vegetation types is due to the applicability of different plant species to the availability of sediment resources, which strongly influences the dynamics of local sediment resources (Zhu et al., 2021). In this study, sediment MBP was found as the largest explanatory factor for the dynamic changes in all sediment P fractions, and showed a significantly positive correlation with all P fractions. MBP is the P contained in the living microorganisms in the sediment (Achat et al., 2010). The correlation between MBP and P fractions in sediments indicates that the P fractions in sediments are mainly driven by biological processes dominated by plant microorganisms (Achat et al., 2010; Fan et al., 2021). Our conclusions are similar to those of some previous studies, indicating that differences in the microbial characteristics of the sediment across different vegetation types may be an essential factor affecting the P fractions. At the same time, combined with the AKP activity, MBP content and the proportion of each P fraction in the surface layer of 0–10 cm sediments, it was found that the enzyme activity and MBP content of the mudflat bare land sediments were significantly lower than those in the sediment under vegetation cover. This further indicates that vegetation regulates the transformation of different P forms in the sediment by influencing microbial bioactivity in rhizosphere sediments, and further determining the bioavailability of P in sediments (Zhong et al., 2021; Fan et al., 2021). In addition, the study also found that the AKP activity and MBP content under the *A. philoxeroides* are significantly higher than those under the native vegetation, indicating that *A. philoxeroides* has high phosphorus utilization efficiency and advantages in maintaining sediment fertility (Dassonville et al., 2008). It should be noted that *A. philoxeroides* is an exotic species, and these advantages may be responsible for its strong adaptability and invasive ability.

4.3. Effects of different growth phases and depths on the P content and fractions of sediment

Some studies have shown that the content and morphological changes of sediment P are jointly affected by the following factors such as geochemical processes, soil parent material (Mage and Porder, 2013), soil age (De Schrijver et al., 2012), climate (Vincent et al., 2014), biological activities (De Feudis et al., 2016) and soil physicochemical properties (Vincent et al., 2011). In this study, the sampling sites were close, and the temporal and spatial changes in sediment P content and fractions caused by differences in soil parent material and age can be ignored. Therefore, we can better compare the effects of different plant growth seasons and depths on the P fractions of sediments in the river-lake ecotone. This study found that the contents of total P and labile P in the sediments increased gradually from the seedling period to the growing period, and decreased gradually from the flowering and fruiting period to the decay period, showing a single-peak type change. The seasonal differences are the result of a combination of biological factors. First, there are different temperatures in different seasons. In this study, the sediment temperature was significantly positively correlated with AKP and MBP, while it was significantly negatively correlated with SOM. Both AKP and MBP are mainly affected by temperature and moisture content (Fang et al., 2015). The temperature gradually increases from the seeding period to the growing period, while the lake enters the rainy season. The lake water moistens the river-lake ecotone, and the sediments reach the appropriate temperature and humidity. Microorganisms rapidly decompose the litter accumulated in winter, and the AKP enzyme also accelerates the promotion of organic phosphate mineralization. (Nannipieri et al., 2011). However, as the gradual decrease in temperature and the arrival of the dry season led to the drying up of wetlands in the river-lake ecotone, plant life history also moved from the flowering and fruiting period to the decay period, which reduced microbial activity and quantity. Secondly, the assimilation of plants is one of the factors affecting the changes in labile phosphorus (Unger et al., 2010). Throughout its life cycle, vegetation continuously absorbs phosphorus nutrients from sediments, and different growing seasons produce litter of varying quantities and quality (Mori et al., 2020). These lead to the redistribution of total phosphorus content and phosphorus fractions in different seasonal sediments (Chen et al., 2021). At the same time, the belowground root activity of plants also varies in different seasons. During spring and summer, the roots grow rapidly and have vigorous activity that can secrete numerous organic acids to reduce the pH of the rhizosphere sediments, activate the non-labile P fraction in the sediments and promote the change of the P fraction by dissolving iron and aluminum oxides in the sediments and forming chelates of iron, aluminum and calcium plasma (Landeweert et al., 2001).

There were significant vertical distribution differences existed in the depth of the P fraction content in the sediment. The total P content in the surface layer (0–10 cm) is the highest, and the P available for plant absorption and utilization decreases with the increase in depth. The main reason for this result is that the surface sediments are most affected by organisms and have more residues from plants and microorganisms than the deeper layers. This part of the decomposition process produces more unstable organic P. A portion of organophosphorus can be converted into bioavailable P through decomposition and mineralization processes (Yuan et al., 2020). Secondly, the reduction of the P content in the lower layer is also due to the leaching of organic acids generated after the decomposition of litter residues into the lower layer, resulting in the release of P in the lower layer. Part of the released Ps is returned to the surface of the sediment (Ippolito et al., 2010). Another reason is the pumping effect of the surface vegetation, which also leads to the continuous transfer of P from the lower layer to the surface layer (Jobbágy and Jackson, 2004). This result can further prove the effect of vegetation restoration on the dynamics of sediment P fractions.

5. Conclusion

In conclusion, the values of total P and bioavailable P in the sediments during the vegetation restoration were significantly higher than those in the unrestored mudflat bare land, indicating vegetation restoration's effect on the migration and transformation of P within the sediments. This change can protect lake water from the threat of eutrophication in the river-lake ecotone. It reflects the ecological function of vegetation and the river-lake ecotone. In addition, there were also significant differences in sediment P fractions among different vegetation types, vegetation growing seasons and sediment depths. This suggests that the ability of plants to influence the P fractions of sediments depends on plant type and growth characteristics. Although the total biomass and P storage per unit area of the invasive plants were the lowest among the four species, the contents of AKP and MBP in the sediments near their roots were the highest among the four species. These results suggest that the efficient utilization of P resources may be one of the secrets of the highly competitive and successful invasion of invasive plants. The different growing seasons and sediment depths reflect the effects of different environmental and biological factors. It is the role of temperature, moisture content, plants, microorganisms and extracellular enzymes in the biogeochemical cycle of sediment P. Through these fundamental studies, vegetation restoration and promoting the sediment P cycle can be better selected to improve the ecological functions of wetlands in the degraded river-lake ecotone in the process of ecological restoration.

CRediT author statement

Zhiwei Yan: Conceptualization, Methodology, Software, Writing - Original Draft, Visualization, Investigation; **Ling Wu:** Investigation, Software; **Tian Lv:** Software, revisions; **Chao Tong, Zhongyao Gao, Yuan Liu** and **Bin Xing:** Investigation; **Chuanxin Chao, Yang Li** and **Ligong Wang:** Methodology; **Dan Yu:** Project administration, Supervision, Funding acquisition; **Chunhua Liu:** Validation, Revision, Funding acquisition. All the authors contributed to editing the manuscript.

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.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2022.119650.

References

Achat, D.L., Morel, C., Bakker, M.R., Augusto, L., Pellerin, S., Gallet-Budynek, A., Gonzalez, M., 2010. Assessing turnover of microbial biomass phosphorus: combination of an isotopic dilution method with a mass balance model. Soil Biol. Biochem. 42, 2231–2240.

Albaladejo, J., Martinez-Mena, M., Roldan, A., Castillo, V., 1998. Soil degradation and desertification induced by vegetation removal in a semiarid environment. Soil Use Manag. 14, 1–5.

Ashraf P, M., Edwin, L., Meenakumari, B., 2006. Studies on the seasonal changes of phosphorus in the marine environments off Cochin. Environ. Int. 32, 159–164.

Cerdà, A., Lucas-Borja, M.E., Franch-Pardo, I., Úbeda, X., Novara, A., López-Vicente, M., Popović, Z., Pulido, M., 2021. The role of plant species on runoff and soil erosion in a Mediterranean shrubland. Sci. Total Environ. 799, 149218.

Chen, X., Condron, L.M., Dunfield, K.E., Wakelin, S.A., Chen, L., 2021. Impact of grassland afforestation with contrasting tree species on soil phosphorus fractions and alkaline phosphatase gene communities. Soil Biol. Biochem. 159, 108274.

Chen, Y., Chen, L., Peng, Y., Ding, J., Li, F., Yang, G., Kou, D., Liu, L., Fang, K., Zhang, B., Wang, J., Yang, Y., 2016. Linking microbial C:N:P stoichiometry to microbial community and abiotic factors along a 3500-km grassland transect on the Tibetan Plateau. Global Ecol. Biogeogr. 25, 1416–1427.

Cleveland, C.C., Liptzin, D., 2007. C:N:P stoichiometry in soil: is there a “Redfield ratio” for the microbial biomass? Biogeochemistry 85, 235–252.

Crews, T.E., Brookes, P.C., 2014. Changes in soil phosphorus forms through time in perennial versus annual agroecosystems. Agric. Ecosyst. Environ. 184, 168–181.

Dassonville, N., Vanderhoeven, S., Vanparys, V., Hayez, M., Gruber, W., Meerts, P., 2008. Impacts of alien invasive plants on soil nutrients are correlated with initial site conditions in NW Europe. Oecologia 157, 131–140.

De Feudis, M., Cardelli, V., Massaccesi, L., Bol, R., Willbold, S., Cocco, S., Corti, G., Agnelli, A., 2016. Effect of beech (Fagus sylvatica L.) rhizosphere on phosphorous availability in soils at different altitudes (Central Italy). Geoderma 276, 53–63.

De Schrijver, A., Vesterdal, L., Hansen, K., De Frenne, P., Augusto, L., Achat, D.L., Staelens, J., Baeten, L., De Keersmaeker, L., De Neve, S., Verheyen, K., 2012. Four decades of post-agricultural forest development have caused major redistributions of soil phosphorus fractions. Oecologia 169, 221–234.

Dodd, R.J., Sharpley, A.N., Berry, L.G., 2018. Organic phosphorus can make an important contribution to phosphorus loss from riparian buffers. Agricult. Environ. Lett. 3, 180002.

Fan, Y., Lu, S., He, M., Yang, L., Hu, W., Yang, Z., Liu, X., Hui, D., Guo, J., Yang, Y., 2021. Long-term throughfall exclusion decreases soil organic phosphorus associated with reduced plant roots and soil microbial biomass in a subtropical forest. Geoderma 404, 115309.

Fang, X., Zhou, G., Li, Y., Liu, S., Chu, G., Xu, Z., Liu, J., 2015. Warming effects on biomass and composition of microbial communities and enzyme activities within soil aggregates in subtropical forest. Biol. Fertil. Soils 52, 353–365.

Fu, D., Wu, X., Duan, C., Chadwick, D.R., Jones, D.L., 2020. Response of soil phosphorus fractions and fluxes to different vegetation restoration types in a subtropical mountain ecosystem. Catena 193, 104663.

Gao, D., Bai, E., Li, M., Zhao, C., Yu, K., Hagedorn, F., 2020. Responses of soil nitrogen and phosphorus cycling to drying and rewetting cycles: a meta-analysis. Soil Biol. Biochem. 148, 107896.

Gao, G., Ding, G., Zhao, Y., Wu, B., Zhang, Y., Qin, S., Bao, Y., Yu, M., Liu, Y., 2014. Fractal approach to estimating changes in soil properties following the establishment of Caragana korshinskii shelterbelts in Ningxia, NW China. Ecol. Indicat. 43, 236–243.

Grafe, M., Goers, M., von Tucher, S., Baum, C., Zimmer, D., Leinweber, P., Vestergaard, G., Kublik, S., Schloter, M., Schulz, S., 2018. Bacterial potentials for uptake, solubilization and mineralization of extracellular phosphorus in agricultural soils are highly stable under different fertilization regimes. Environ. Microbiol. Rep. 10, 320–327.

Hilt, S., Alirangues Nunez, M.M., Bakker, E.S., Blindow, I., Davidson, T.A., Gillefalk, M., Hansson, L.A., Janse, J.H., Janssen, A.B.G., Jeppesen, E., Kabus, T., Kelly, A., Kohler, J., Lauridsen, T.L., Mooij, W.M., Noordhuis, R., Phillips, G., Rucker, J., Schuster, H.H., Sondergaard, M., Teurlincx, S., van de Weyer, K., van Donk, E., Waterstraat, A., Willby, N., Sayer, C.D., 2018. Response of submerged macrophyte communities to external and internal restoration measures in north temperate shallow lakes. Front. Plant Sci. 9, 194.

Hongthanat, N., Kovar, J.L., Thompson, M.L., Russell, J.R., Isenhart, T.M., 2016. Phosphorus source—sink relationships of stream sediments in the Rathbun Lake watershed in southern Iowa, USA. Environ. Monit. Assess. 188, 453.

Hou, E., Luo, Y., Kuang, Y., Chen, C., Lu, X., Jiang, L., Luo, X., Wen, D., 2020. Global meta-analysis shows pervasive phosphorus limitation of aboveground plant production in natural terrestrial ecosystems. Nat. Commun. 11, 637.

Ippolito, J.A., Blecker, S.W., Freeman, C.L., McCulley, R.L., Blair, J.M., Kelly, E.F., 2010. Phosphorus biogeochemistry across a precipitation gradient in grasslands of central North America. J. Arid Environ. 74, 954–961.

Jenkinson, D., 2004. Measuring soil microbial biomass. Soil Biol. Biochem. 36, 5–7.

Jin, Z., Luo, D., Yu, Y., Yang, S., Zhang, J., Cao, G., 2022. Soil pH changes in a small catchment on the Chinese Loess Plateau after long-term vegetation rehabilitation. Ecol. Eng. 175, 106503.

Jobbágy, E.G., Jackson, R.B., 2004. The uplift of soil nutrients by plants: biogeochemical consequences across scales. Ecology 85, 2380–2389.

Johnson, L.T., Baker, D.B., Confesor, R.B., Krieger, K.A., Richards, R.P., 2014. Research to help lake erie: proceedings of the “phosphorus along the land–river–lake continuum” research planning and coordination workshop. J. Great Lake. Res. 40, 574–577.

Krishna Prasad, M.B., Ramanathan, A.L., 2008. Sedimentary nutrient dynamics in a tropical estuarine mangrove ecosystem. Estuar. Coast Shelf Sci. 80, 60–66.

Lai, J., Zou, Y., Zhang, J., Peres Neto, P.R., 2022. Generalizing hierarchical and variation partitioning in multiple regression and canonical analyses using the rdacca.hp R package. Methods Ecol. Evol. 13, 782–788.

Landeweert, R., Hoffland, E., Finlay, R.D., Kuyper, T.W., van Breemen, N., 2001. Linking plants to rocks: ectomycorrhizal fungi mobilize nutrients from minerals. Trends Ecol. Evol. 16, 248–254.

Li, B., Brett, M.T., 2013. The influence of dissolved phosphorus molecular form on recalcitrance and bioavailability. Environ. Pollut. 182, 37–44.

Li, Y., Wang, L., Chao, C., Yu, H., Yu, D., Liu, C., 2021. Submerged macrophytes successfully restored a subtropical aquacultural lake by controlling its internal phosphorus loading. Environ. Pollut. 268, 115949.

Luo, L., Ye, H., Zhang, D., Gu, J.D., Deng, O., 2021. The dynamics of phosphorus fractions and the factors driving phosphorus cycle in Zoige Plateau peatland soil. Chemosphere 278, 130501.

Mage, S.M., Porder, S., 2013. Parent material and topography determine soil phosphorus status in the luquillo mountains of Puerto Rico. Ecosystems 16, 284–294.

Mäkiranta, P., Laiho, R., Mehtätalo, L., Straková, P., Sormunen, J., Minkkinen, K., Penttilä, T., Fritze, H., Tuittila, E.S., 2018. Responses of phenology and biomass production of boreal fens to climate warming under different water-table level regimes. Global Change Biol. 24, 944–956.

Maranguit, D., Guillaume, T., Kuzyakov, Y., 2017. Land-use change affects phosphorus fractions in highly weathered tropical soils. Catena 149, 385–393.

Mori, A.S., Cornelissen, J.H.C., Fujii, S., Okada, K., Isbell, F., 2020. A meta-analysis on decomposition quantifies afterlife effects of plant diversity as a global change driver. Nat. Commun. 11, 4547.

Nannipieri, P., Giagnoni, L., Landi, L., Renella, G., 2011. Role of phosphatase enzymes in soil. In: Bünemann, E., Oberson, A., Frossard, E. (Eds.), Phosphorus in Action: Biological Processes in Soil Phosphorus Cycling. Springer Berlin Heidelberg, Berlin, Heidelberg.

Norgbey, E., Li, Y., Zhu, Y., Nwankwegu, A.S., Bofah-Buah, R., Nuamah, L., 2021. Seasonal dynamics of iron and phosphorus in reservoir sediments in Eucalyptus plantation region. Ecol. Process. 10.

Ovsepyan, L., Kurganova, I., Lopes De Gerenyu, V., Kuzyakov, Y., 2020. Conversion of cropland to natural vegetation boosts microbial and enzyme activities in soil. Sci. Total Environ. 743, 140829.

Pang, J., Ryan, M.H., Tibbett, M., Cawthray, G.R., Siddique, K.H.M., Bolland, M.D.A., Denton, M.D., Lambers, H., 2009. Variation in morphological and physiological parameters in herbaceous perennial legumes in response to phosphorus supply. Plant Soil 331, 241–255.

Rodrigues, M., Pavinato, P.S., Withers, P.J., Teles, A.P., Herrera, W.F., 2016. Legacy phosphorus and no tillage agriculture in tropical oxisols of the Brazilian savanna. Sci. Total Environ. 542, 1050–1061.

Schindler, D.W., Carpenter, S.R., Chapra, S.C., Hecky, R.E., Orihel, D.M., 2016. Reducing phosphorus to curb lake eutrophication is a success. Environ. Sci. Technol. 50, 8923–8929.

Su, Y., Zhao, H.l., 2003. Soil properties and plant species in an age sequence of Caragana microphylla plantations in the Horqin Sandy Land, north China. Ecol. Eng. 20, 223–235.

Tang, X., Li, R., Han, D., Wu, X., 2020. Impacts of electrokinetic isolation of phosphorus through pore water drainage on sediment phosphorus storage dynamics. Environ. Pollut. 266, 115210.

Tiessen, H., Moir, J.O., 1993. Characterization of available P by sequential extraction. Soil Sampl. Methods Anal. 75–86.

Unger, M., Leuschner, C., Homeier, J., 2010. Variability of indices of macronutrient availability in soils at different spatial scales along an elevation transect in tropical moist forests (NE Ecuador). Plant Soil 336, 443–458.

Vincent, A.G., Schleucher, J., Gröbner, G., Vestergren, J., Persson, P., Jansson, M., Giesler, R., 2011. Changes in organic phosphorus composition in boreal forest humus soils: the role of iron and aluminium. Biogeochemistry 108, 485–499.

Vincent, A.G., Sundqvist, M.K., Wardle, D.A., Giesler, R., 2014. Bioavailable soil phosphorus decreases with increasing elevation in a subarctic tundra landscape. PLoS One 9, e92942.

Vitousek, P.M., Porder, S., Houlton, B.Z., Chadwick, O.A., 2010. Terrestrial phosphorus limitation: mechanisms, implications, and nitrogen–phosphorus interactions. Ecol. Appl. 20, 5–15.

Wang, C., Xue, L., Jiao, R., 2021. Soil phosphorus fractions, phosphatase activity, and the abundance of phoC and phoD genes vary with planting density in subtropical Chinese fir plantations. Soil Tillage Res. 209, 104946.

Wang, W., Sardans, J., Wang, C., Zeng, C., Tong, C., Chen, G., Huang, J., Pan, H., Peguero, G., Vallicrosa, H., Penuelas, J., 2019. The response of stocks of C, N, and P to plant invasion in the coastal wetlands of China. Global Change Biol. 25, 733–743.

Xiao, J., Xie, B., Zhou, K., Shi, S., Li, J., Yang, M., Liu, C., 2021. Assessment of soil erosion in the Dongting Lake basin, China: patterns, drivers, and implications. PLoS One 16, e0261842.

Yan, Z., Wang, Q., Li, Y., Wu, L., Wang, J., Xing, B., Yu, D., Wang, L., Liu, C., 2021. Combined effects of warming and nutrient enrichment on water properties, growth, reproductive strategies and nutrient stoichiometry of Potamogeton crispus. Environ. Exp. Bot. 190, 104572.

Yuan, H., Tai, Z., Li, Q., Zhang, F., 2020. Characterization and source identification of organic phosphorus in sediments of a hypereutrophic lake. Environ. Pollut. 257, 113500.

Zhang, Y., Xu, X., Li, Z., Liu, M., Xu, C., Zhang, R., Luo, W., 2019. Effects of vegetation restoration on soil quality in degraded karst landscapes of southwest China. Sci. Total Environ. 650, 2657–2665.

Zhao, Q., Zeng, D., Fan, Z., Yu, Z., Hu, Y., Zhang, J., 2009. Seasonal variations in phosphorus fractions in semiarid sandy soils under different vegetation types. For. Ecol. Manag. 258, 1376–1382.

Zhong, H., Smith, C., Robinson, B., Kim, Y.N., Dickinson, N., 2021. Soil phosphorus dynamics along a short-term ecological restoration trajectory of a coastal sandplain forest in New Zealand. Land Degrad. Dev. 32, 1250–1261.

Zhou, J., Wu, Y., Bing, H., Yang, Z., Wang, J., Sun, H., Sun, S., Luo, J., 2016. Variations in soil phosphorus biogeochemistry across six vegetation types along an altitudinal gradient in SW China. Catena 142, 102–111.

Zhu, X., Fang, X., Wang, L., Xiang, W., Alharbi, H.A., Lei, P., Kuzyakov, Y., 2021. Regulation of soil phosphorus availability and composition during forest succession in subtropics. For. Ecol. Manag. 502.