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Redeposited particles mitigate dredging effect on internal phosphorus loading by supplying phosphorus and fueling sediment phosphorus cycling

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Abstract

Sediment dredging is widely applied to mitigate sediment P loading in hypereutrophic systems, yet its long-term effectiveness under dynamic particle deposition remains poorly constrained. In this study, a one-year field incubation in Lake Taihu combined high-resolution porewater solute profiling, benthic chemical flux measurements, microbial network analysis, sediment characterization, and phosphate oxygen isotope ($\delta^{18}\text{O}_\text{P}$) tracing to investigate sediment P biogeochemistry following simulated-dredging with/without settling particle inputs. Dredging suppressed porewater SRP and Fe^{2+} accumulation, and benthic SRP flux, thereby limiting sediment P remobilization in the absence of settling particles. However, under continuous particle deposition, sediment P cycling was rapidly re-established, particularly during warm seasons. Notably, benthic SRP efflux followed a power-law relationship with sediment oxygen uptake ($F_{\text{SRP}}=0.542\text{SOU}^{0.410}$, $r^2 = 0.93$), indicating additional P contributions from redox-mediated feedbacks, such as Fe-P remobilization, beyond organic matter remineralization alone. Microbial and phosphate solubilizing bacteria (PSB) networks were simplified yet more stable after dredging, but particle inputs weakened this effect, reinforcing coupled biogeochemical cycling. $\delta^{18}\text{O}_\text{P}$ fingerprints further revealed that post-dredging sediment recovery was primarily fueled by redeposition of re/suspended local particles rather than nonlocal inputs. Dredging triggers an interruption of intralake P recycling but should be applied alongside complementary geoengineering to optimize eutrophication management.

Keywords: Sediment dredging; Legacy phosphorus; Phosphate oxygen isotope ($\delta^{18}\text{O}_\text{P}$); Settling particles (SP); Microbial network

1. Introduction

Phosphorus (P) is an essential nutrient for aquatic primary productivity, yet also a primary driver of eutrophication in freshwater ecosystems (Li et al., 2017a; Duhamel et al., 2021). Elevated P availability, derived from both external inputs and internal sediment recycling, can sustain degraded ecological states and threaten drinking-water security and aquatic ecosystem health (Paerl et al., 2016). In eutrophic lakes, sediment P loading often becomes the dominant source maintaining eutrophication even after external inputs are readily managed (Jeppesen et al., 2005; Katsev and Dittrich, 2013).

Sediment P loading originates mainly from the deposition of organic matter (OM) and P adsorbed onto redox-sensitive metallic oxides, particularly iron oxy/hydroxides, derived from the overlying productive water column (Li et al., 2018). The redistribution of sedimented P, either recycled back into the water column or retained as refractory forms at depth, is predominantly governed by microbially mediated diagenetic processes (Burdige, 2006). Upon deposition of OM and particle-sorbed P onto the surface sediment, heterotrophic bacteria oxidize OM sequentially using electron acceptors such as oxygen, nitrate, Mn/Fe oxides, and sulfate for metabolism, resulting in the mineralization of organic P and the reductive dissolution of metal-bound P into the ambient porewater. This establishes a positive porewater soluble reactive phosphate (SRP) concentration gradient that drives SRP efflux across the sediment water interface (SWI). In the presence of iron oxyhydroxides, specifically ferrihydrite, in the surface oxic layer, a redox-sensitive P barrier is formed which can immobilize SRP sourced from OM mineralization and diffusion from anoxic layers at depth; this suppresses SRP efflux under well-oxygenated conditions but exponentially amplifies benthic P release once oxygen deficiency occurs (Münch et al., 2024). P can also be sequestered by calcium and/or aluminum hydroxides to form inert P species, or coprecipitate with oversaturated Fe^{2+} in the anoxic layers as vivianite, prompting permanent P burial (Jilbert and Slomp, 2013; Slomp, 2023). This tightly coupled C, Fe, and P interaction underscores the sensitivity of P biogeochemical cycling to redox conditions and OM availability.

Sediment dredging has been widely applied as a reliable geoengineering approach to mitigate sediment P loading by removing OM-rich surface sediments and altering the microbial processes (Kleeberg and Kohl, 1999; Wen et al., 2020). While short-term reductions in P release are frequently observed, long-term effectiveness remains inconsistent (Yang et al., 2024); for example, a fifteen-year monitoring in a sublake of Lake Taihu has reported a reversed benthic P release due to intensive settling particle (SP) inputs just 18 months after dredging (Liu et al., 2016). A critical but insufficiently resolved factor is the role of post-dredging particle dynamics (Søndergaard et al., 2007; Lüring and Faassen, 2012). In productive lakes, settling particles (SP) introduces substantial labile OM, reactive Fe and P, as well as microorganisms, potentially reactivating sediment biogeochemical processes and undermining dredging effectiveness (Huser et al., 2016; Liu et al., 2022). Despite its recognised importance, the source identification of SP inputs, the resulting microbial responses and their biogeochemical consequences, remain poorly constrained in dredging-remediated systems.

Phosphate oxygen isotopes ($\delta^{18}\text{O}_\text{P}$) provides a robust tool to elucidate P sources and cycling pathways, as the strong phosphorus-oxygen (P-O) bond preserves isotopic signatures during abiotic processes such as transport, sorption/desorption, and precipitation, under most earth environments (Jaisi and Blake, 2014; Li et al., 2017a; Wang et al., 2023). This enables differentiation between locally recycled and externally derived P sources in post-dredging systems (Jaisi et al., 2011). Depending on the specific environmental conditions of different water bodies, sediment P content may increase to varying degrees after dredging. Identifying the sources of P in post-dredging sediments is crucial for maintaining dredging effects and developing targeted management approaches.

In this study, we investigated sediment P biogeochemistry and its origins following simulated-dredging under contrasting particle deposition scenarios via a one-year in situ incubation experiment, which captures the typical redox environments in Lake Taihu. We quantitatively assessed the effects of sediment dredging on coupled C-Fe-P

cycling, sediment redox conditions and SRP fluxes at the SWI, and microbial community dynamics under scenarios with and without particle deposition. Furthermore, we identified the sources of SP using the phosphate oxygen isotope ($\delta^{18}\text{O}_\text{P}$) technique. Our study provides new mechanistic insights into post-dredging sediment recovery and offers guidance for improving long-term eutrophication management.

2. Materials and methods

2.1 Sampling and simulated-dredging experiment strategies

The sampling site is situated in the inner Meiliang Bay in northern Lake Taihu ($31^\circ 31' 33.9''$ N, $120^\circ 12' 35.2''$ E, [Fig S1](#)). This region has suffered from persistent eutrophication and frequent harmful algal blooms driven primarily by excessive terrestrial pollutant inputs and hydrodynamically induced algal migration and accumulation ([Qin et al., 2007](#)). Consequently, persistent algal and particle deposition has resulted in severe sediment internal nutrient loading in this relatively shallow area ($\sim 1.5\text{m}$). Sediment dredging has been proposed by the local environmental management agency as a strategy to mitigate internal loading.

In this simulated-dredging experiment, a total of 42 intact sediment cores (50cm depth) were collected using a hand-hold corer in January 2021. Upon retrieval, three treatments were established as follows: UDS and DS treatments were the topmost 20cm and 30-50cm depth as the undredged and simulated-dredging sediment exposed to SP influence, respectively; D treatment was covered with permeable 800-mesh polyamide to diminish SP inputs (although potential effects on fauna activities, hydrodynamics, and diffusive boundary-layer near the SWI cannot be excluded). The polyamide mesh was replaced monthly to minimize clogging and maintain water and oxygen transfer. This simulated-dredging depth of 30cm was selected to reflect local dredging practices ([Yu et al., 2016](#)). Additionally, triplicate open topped tubes (86mm ID and 20cm length) were deployed to collect deposited SP. All prepared cores were fixed onto stainless-steel frames in pairs and were placed onto the lakebed ~ 100 m from shore at the Lake Taihu Ecosystem National Field Scientific Observation Station, Chinese Academy of Science ($31^\circ 25' 10.78''$ N, $120^\circ 12' 50.7''$ E). This field incubation allowed the cores to

be exposed to natural lake conditions.

After sampling, triplicate sediment cores of each treatment and collected SP were sacrificially retrieved bimonthly at 0d (January 2021), 60d (March), 120d (May), 180d (July), 240d (September), 300d (November), and 360d (January 2022). Simultaneously, the physicochemical properties of the water column, including water temperature, pH, dissolved oxygen (DO), and chlorophyll. a (Chl. a) concentration, were measured using a portable multiparameter meter (YSI, Yellow Springs, USA), and 2L of in-situ overlying water was collected using a Niskin bottle in a pre-cleaned polyethylene bottle for subsequent lab analysis. Additionally, 60L of overlaying water, sediment and soil samples from the surface sediment at the incubation site, and lakeshore, respectively, were collected for phosphate oxygen isotope ($\delta^{18}\text{O}_\text{P}$) analysis at 180d. All samples were transported to the lab within hours.

For laboratory analysis, each short sediment core was first extruded upward into a long PC tube (86mm ID and 50cm length) without disturbance. Then, 0.8 μm -filtered in-situ overlying water was carefully added dropwise along the tube wall to avoid sediment resuspension until 20 cm depth was obtained, and the cores were then stabilized overnight to allow the sediment-water boundary layer to re-establish. Each core was then sequentially used for DO micro-profile, incubated SRP fluxes, porewater SRP and Fe^{2+} , and DGT-labile P concentration measurements. Finally, these sediment cores were sectioned at 2 cm intervals to the bottom for further biological properties and solid phase analysis.

2.2 Sediment porewater dissolved O_2 , SRP, and Fe^{2+} measurements

Sediment porewater DO micro-profiles were measured instantly after retrieval using a Clark-type O_2 microsensor (Unisense, Denmark), which was pre-calibrated using air-saturated deionized water and an oversaturated sodium dithionite solution as the deoxygenated point under in-situ temperature.

Sediment porewater SRP and Fe^{2+} , and DGT-labile P concentrations were determined using high-resolution passive equilibrium samplers (HR-Peeper, 4mm resolution) and ZrO-AT DGT probes (Easysenor Ltd., China), respectively, after intact

core incubation experiments. In each core, HR-Peeper and DGT probe were inserted vertically into the sediment and allowed to equilibrate for up to 48h and 24h, respectively. Upon retrieval, porewater was immediately pipetted into a microplate for SRP and Fe^{2+} photometric analysis using the molybdenum blue and Ferrozine methods, respectively. The DGT binding gels were sliced at 2 mm intervals, and P was eluted using $1 \text{ mol} \cdot \text{L}^{-1}$ NaOH for 16h. The DGT-labile P concentrations were calculated using Eq. (1); for detailed deployment and DGT Induced Fluxes in Sediments (DIFS) model information please see S1 in Supplementary Materials:

$$C_{DGT} = \frac{M \times \Delta_g}{D \times A \times t} \quad (1)$$

where M is the accumulated DGT-labile P on the binding gel (mg); Δ_g is the thickness of the diffusive gel layer (cm); D is the diffusion coefficient calibrated by temperature, detailed values please refer to Wang et al., (2016); A is the exposed surface area of the DGT probe; and t is the deployment time (s).

2.3 Estimation of SRP, Fe^{2+} , and O_2 fluxes across the SWI

In this study, the molecular diffusive flux and incubated flux were estimated using the porewater dissolved species gradient at the SWI and the intact-core incubation experiment, respectively. The molecular diffusive fluxes (F_i^{diff}) of the dissolved SRP, DO, and Fe^{2+} were calculated using Fick's first law of diffusion, as described in Eq. (2):

$$F_i^{\text{diff}} = \varphi D_i^s \frac{dC_i}{dz} \quad (2)$$

where F_i^{diff} is the diffusive flux of the interest i, e.g., SRP ($\text{mg} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$); φ is the porosity of the surface sediment; D_i^s is the molecular diffusivity corrected for in-situ temperature and surface sediment tortuosity (Table S1); for SRP, we used the diffusion coefficient of the dominant HPO_4^{2-} phosphate in circumneutral freshwater lake (Boudreau, 1997); C is the porewater or overlying water concentrations ($\text{mg} \cdot \text{L}^{-1}$); and z is the diffusion depth (cm). Positive fluxes indicate solute release from the sediment (efflux), whereas negative fluxes indicate solute uptake by the sediment (influx).

For extra contributions beyond molecular diffusion, apparent total SRP fluxes were measured using an intact-core incubation experiment (please see detailed

procedures in S2 in Supplementary Material and our previous work) (Sun et al., 2022).

The total SRP flux can be calculated using Eq. (3):

$$F_{incubation} = [V(C_n - C_0 + \sum_{j=1}^n V_{j-1}(C_{j-1} - C_a))]/(S \times t) \quad (3)$$

where $F_{incubation}$ is the averaged total SRP flux across the SWI over three days, ($\text{mg} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$); V is the volume of the water column (L^{-1}); C_n , C_0 , and C_{j-1} are the SRP concentrations at n , 0 , and $j-1$ time ($\text{mg} \cdot \text{L}^{-1}$), respectively; C_a is the SRP concentration in the in-situ overlying water ($\text{mg} \cdot \text{L}^{-1}$); V_{j-1} is the volume of sampled overlying water each time (L^{-1}); S is the area of the SWI (m^2); t is the incubation time (d^{-1}).

2.4 Analysis of sediment biological properties

Microbial total activity and alkaline phosphatase activity (APA) were measured using fresh wet surface sediment (0-2 cm) and SP subsamples, following methods described in Adam and Duncan, (2001) and Adams et al., (2008), respectively. Detailed protocols are provided in S3 and S4 in Supplementary Material.

Sediment microbial DNA information was extracted from freeze-dried sediments following the protocol of Fast DNATM SPIN Kit for Soil (MP biomedical, USA) (see S5 and S6 in Supplementary Material). The quantified products were measured by a commercial company (Biozeron Co., Ltd, Shanghai, China) on an Illumina MiSeq platform, and raw sequencing data was uploaded to the NCBI SRA database under the accession number: PRJNA1455951.

2.5 Analysis of sediment solid properties

Sediment water content and porosity were determined from moisture loss after freeze-drying under vacuum using a cutting-ring method. Freeze-dried sediments were ground, homogenized, and sieved through 0.15mm mesh for subsequent analysis. Loss on ignition (LOI) was determined as the mass loss after ignition at 550°C for 5h to represent sediment organic matter content.

Solid sediment subsamples were homogenized for P fractionation using a P sequential extraction technique (Table S2). This method operationally separates different P fractions in order of their availability, including loosely sorbed labile P

(NH₄Cl-P), redox-sensitive P bound to Fe and/or Mn oxyhydroxides (Fe-P), organic P (Org-P), aluminum bound P (Al-P), apatite bound P (Ca-P), and refractory organic and inorganic P (Res-P). Meanwhile, surface sediment P adsorption ability was quantified using the phosphate adsorption isotherm experiment as described by [Sun et al., \(2023\)](#), the quantitative parameters, e.g., $Q_{\max}$ (the maximum P adsorption capacity, mg·kg⁻¹), and EPC₀ (zero equilibrium P concentration, mg·L⁻¹), were calculated using the non-linear Langmuir equation. All SRP concentrations in sediment extracts were determined using the molybdate blue method described above.

Additionally, phosphate oxygen isotopes were measured after purifying water samples or using dry sediment samples. Detailed pre-treatment protocols were adapted to [Jiang et al., \(2017\)](#). Isotope measurements were performed using a High Temperature Conversion - Isotope Ratio Mass Spectrometer (Flash EA 1112 Series, Delta V Advantage, Thermo Fisher Scientific), and results were reported relative to the VSMOW standard.

2.6 Sequence and Statistical Analyses

3,446,938 raw paired-end reads were assembled using FLASH v1.2.7 and processed with the QIIME v.1.8.0 pipeline with the following settings: sequences of length < 200 or > 500, average quality < 20, ambiguous bases > 0 or homopolymer length > 6 were removed. After quality filtering, 2,986,342 reads remained and chimeric sequences were identified and removed by UCHIME. The remaining 2,845,635 sequences were clustered into operational taxonomic units (OTUs) at a 97% similarity using the UPARSE. Then the OTUs with sequences number lower than 3 were removed. Representative sequences were taxonomically assigned against the SILVA 138 dataset using the RDP classifier v.2.2. To standardize sequencing depth, all samples were rarefied to 26,161 sequences.

Alpha diversity indices, including richness, Shannon-Wiener index, and evenness, were calculated using the vegan package in R v4.0.4. Differences among groups were tested using Kruskal-Wallis tests followed by pairwise Wilcoxon tests. Beta diversity was assessed using principal coordinates analysis (PCoA) and permutational

multivariate analysis of variance (PERMANOVA). Prior to PERMANOVA, multivariate homogeneity of group dispersions was tested using the PERMDISP based on Bray-Curtis dissimilarities. PERMANOVA was performed using the `adonis2` function, and pairwise comparisons between groups were performed using the `'pairwise.adonis2'` function of the `pairwiseAdonis` package with 9999 permutations. To correct for multiple testing, p-values from all pairwise comparisons were adjusted using the Benjamini-Hochberg approach. Physicochemical properties, except pH, were log (x+1)-transformed before multivariate analysis. Redundancy analysis (RDA) was performed to examine relationships between microbial community structures and environmental variables. Mantel tests were performed to evaluate correlations between microbial community dissimilarity and environmental factors, and Spearman correlation analysis was used to assess relationships among environmental factors.

Microbial co-occurrence networks were constructed for UDS, DS, D, and SPM samples from 60d to 360d, respectively. For each network, the OTUs that appeared in more than half of the samples were retained for analysis. Pairwise Spearman correlations among OTUs were calculated using the R `Hmisc` package, and only robust and significant correlations ($|r| > 0.6$, $p < 0.01$) were retained. The p-values from all were adjusted using the Benjamini-Hochberg approach (Benjamini and Hochberg, 1995). Moreover, 1000 Erdős-Rényi random networks were generated for comparison with the topology of a real network, with each edge possessing an equal probability of being assigned to any node (Erdős and Rényi, 1960). Networks were visualized in Gephi, and node- and network-level topological parameters were calculated using the R `igraph` package. The network hubs ($Z_i > 2.5$, $P_i > 0.62$), module hubs ($Z_i > 2.5$, $P_i < 0.62$), connectors ($Z_i < 2.5$, $P_i < 0.62$), and peripherals ($Z_i < 2.5$, $P_i < 0.62$) were identified based on metabolic network approaches. Network stability was evaluated as robustness, defined as the proportion of remaining taxa after random or targeted node removal.

3. Results

3.1 Sediment redox conditions and porewater P-Fe dynamics

Mean oxygen penetration depth (OPD) were 3.87 ± 2.11 (UDS), 4.48 ± 1.99 (DS), and 6.46 ± 1.32 mm (D), demonstrating that dredging significantly ($p < 0.05$) increases OPD (Fig. 1c). OPD was negatively correlated with water-column Chl. a ($r = -0.62$, $p < 0.05$) and sediment Org-P ($r = -0.64$, $p < 0.01$) (Table S3 and Fig. 2). DO levels at the SWI were positively correlated with water-column DO ($r = 0.53$, $p < 0.05$) and negatively correlated with sediment LOI ($r = -0.54$, $p < 0.05$) (Fig. 1b). Improved DO levels were detected in the D treatment.

Porewater SRP and Fe^{2+} profiles generally followed similar vertical patterns (Fig. 3). Porewater Fe^{2+} typically emerged below the OPD, except at 180d, and generally formed subsurface peaks, except for the D treatment. Porewater SRP displayed a similar vertical pattern to Fe^{2+} ($r = 0.78$, $p < 0.001$), and no significant difference was detected between UDS and DS treatments. Porewater SRP was positively correlated with sediment LOI ($r = 0.51$, $p < 0.05$), Org-P content ($r = 0.42$, $p < 0.05$), and Fe-P ($r = 0.52$, $p < 0.05$) (Fig. 2), but negatively related to DO at the SWI ($r = -0.52$, $p < 0.05$), and to DO ($r = -0.65$, $p < 0.01$) and NO_3^- -N ($r = -0.57$, $p < 0.01$) in the overlying water. Notably, negligible SRP and DGT-labile P concentrations were generally observed within Fe^{2+} -depleted scales, except at 180d and 240d (Fig. 3).

Seasonal variability, especially water temperature, significantly influenced porewater SRP ($r = 0.72$, $p < 0.001$) and Fe^{2+} concentrations and gradients in the UDS and DS treatments (Fig. 2). During cold, well-oxygenated seasons (e.g., 60d and 360d), negligible porewater SRP and Fe^{2+} concentrations were measured in the profiles (Fig. 3a and b), and the sediment even acted as a temporary P sink, as supported by SRP influxes, e.g., $-2.326 \pm 0.053 \text{ mg m}^{-2} \text{ d}^{-1}$ in the DS treatment at 60d (Table 1). In contrast, during warm seasons with low-oxygen availability (e.g., 180d and 240d), porewater SRP and Fe^{2+} , bioavailable P (DGT-labile P) concentrations, and P resupply ratios drastically increased, with steepened gradients across the SWI (Fig. 3 and Table S4). The D treatment without SP deposition maintained consistently low porewater SRP and Fe^{2+} concentrations, weak gradients and resupply capacity (R ratio) throughout the study (Fig. 3 and Table S4). Org-P content in the D treatment fluctuated only from 37.47

to 87.32 mg g⁻¹, compared with 77.80 to 206.12 mg g⁻¹ in the DS treatment (Fig. 4a).

3.2 Benthic chemical fluxes and sublinear SOU-SRP response after dredging

As shown in Table 1, we quantified SRP incubation fluxes and chemical (SRP, Fe²⁺, and O₂) diffusive fluxes in this study. SRP incubation fluxes were consistently larger than diffusive fluxes and were positively correlated with diffusive fluxes ($r = 0.776$, $p < 0.001$) (Fig. 5a). Incubated and diffusive SRP fluxes were positively correlated with porewater SRP and Fe²⁺ concentrations ($p < 0.001$), Fe²⁺ diffusive flux ($p < 0.01$), and sediment mobile P pool (NH₄Cl-P, Fe-P, and Org-P) ($p < 0.05$), but negatively correlated with O₂ diffusive flux ($p < 0.05$) (Fig. 2).

Source-sink transition was observed in all three treatments during the cold-to-warm season shift. In cold seasons, the overlying water was highly oxygenated and oxygen penetrated deep into the sediment. SRP diffusive effluxes were linearly related to sediment oxygen uptake (SOU, expressed as the positive magnitude of O₂ diffusive influx), and the slope was generally lower than the Redfield ratio (138O₂: 106C: 1P) (Fig. 5a). Apparent P recycling ratios were generally low (< 1), and negative SRP fluxes were also observed. Meanwhile, BD-P content generally increased in the DS and D treatments during the cold season; for example, Fe-P content in the DS treatment increased from 61.77 to 84.79 mg/kg from 60d to 120d (Fig. 4a). In contrast, P effluxes and apparent P recycling ratios (> 1) drastically increased during warm seasons. During warm seasons, SRP efflux increased with SOU in a sublinear power-law form in the UDS and DS treatments (Fig. 5a, $F_{\text{SRP_flux}} = 0.542 \cdot \text{SOU}^{0.410}$, exponent = 0.410, $r^2 = 0.930$). Enhanced SP deposition increased surface sediment OM availability and different P fractions relative to the D treatment; Org-P and Fe-P in the DS treatment increased from 49.55 and 71.25 mg kg⁻¹ at 0d to 206.12 and 121.36 mg kg⁻¹ at 180d, respectively (Fig. 4a and Fig. S2). Higher O₂ influx, elevated microbial total activity and APA activity were also observed during this period (Fig. S3). Q_{max} values in the DS treatment decreased from 1407.3 mg kg⁻¹ at 60d to 1152.2 mg kg⁻¹ at 180d (Table S5 and Fig. S4). SRP and Fe²⁺ diffusive fluxes were positively correlated in the UDS and DS treatments, with $r^2 = 0.920$ and 0.335, respectively (Fig. 5b). Reduced NO₃⁻-N

availability was also observed in the overlying water during warm seasons (Table S3).

In contrast, similar warm-season amplifying phenomena were not evident in the D treatment. Chemical fluxes across the SWI remained negligible compared with the UDS and DS treatments. After one year of incubation, apparent P recycling ratios were 0.79, 0.09, and 0.00 in the UDS, DS, and D treatments, respectively, and the D treatment maintained lower P recycling after 240d without SP inputs (Table 1).

3.3 Microbial community structures and co-occurrence patterns

In this study, the sequence number ranged from 35,701 to 71,424 (Table S6), 8,905 OTUs (55.22%) were shared across all groups, and the proportion of unique OTUs was lowest in the UDS (10.28%) and highest in the SP group (20.69%) (Fig. 6a). Microbial richness in the DS treatment was significantly lower than that in the UDS treatment, whereas the evenness and Shannon-Wiener index in the D treatment were higher than those in the UDS and DS treatments (Fig. 6b). There were also significant differences in microbial community structures ($n = 72$, pseudo- $F = 6.06$, $R^2 = 0.21$, $p < 0.05$), with the highest and lowest community dissimilarity measured in the UDS and D groups, respectively (Fig. 6c and d). Microbial richness showed similar seasonal patterns in the UDS and SP groups but a distinct temporal lag in the DS group.

The Actinobacteria (average relative abundance: 20.2%), Chloroflexi (17.7%), and Gammaproteobacteria (14.2%) were the dominant taxa in the co-occurrence networks of the UDS, DS, and SP groups, whereas Firmicutes (17.5%) replaced Actinobacteria in the D group (Fig. 7a). The modularity (MD, 0.679–0.882), average shortest path length (APL, 3.858–5.789), and clustering coefficient (CC, 0.30–0.351) of the UDS, DS, D, and SP microbial communities were higher than those of their associated Erdős-Rényi random networks (Table S7). The network in the D treatment was characterized by fewer abundant modules (relative abundance $>1\%$), lower numbers of nodes and edges, and reduced degree and betweenness centrality, but higher closeness centrality than the other treatments (Fig. 7b and Table S7). Network robustness was highest in the D treatment but lowest in the UDS treatment (Fig. 6c).

In addition, phosphate-solubilizing bacteria (PSB) communities were

characterized, with Burkholderia, Sphingomonas, Flavobacterium, Bacillus, Acinetobacter, Thiobacillus, Pseudomonas, Dechloromonas, Paenibacillus, and Brevibacillus as the dominant taxa (Fig. S5a). Acinetobacter was dominant in both the UDS and DS treatments, whereas Burkholderia was enriched in the DS treatment. In contrast, a high proportion of Flavobacterium was detected in SP but was low in the DS and D groups. The PCoA analysis showed significant differences in PSB community composition among treatments (Fig. S5b). The RDA further showed significant relationships between PSB communities and ambient environmental variables (Fig. S5c); for instance, OPD was significantly correlated with the abundances of Thiobacillus, Pseudomonas, and Dechloromonas.

As shown in Fig. S6a and Table S8, Chloroflexi–Chloroflexi and Actinobacteria–Actinobacteria correlations and keystones were dominant in the UDS and DS treatments, respectively, whereas Firmicutes–Firmicutes correlations dominated in the D treatment. Meanwhile, the UDS treatment contained more keystone taxa with diverse functions (36) than the DS (12) and D (4) treatments (Table S8), including taxa associated with anaerobic fermentation (Anaerolineaceae), nitrification (Nitrosomonadaceae), and sulfate reduction (Desulfosarcinaceae). In comparison, keystone taxa in the DS and D treatments were mainly associated with sulfur-containing OM degradation (Hydrogenophilaceae) and dissimilatory iron reduction (Paludibaculum), respectively. Environmental variables showed the strongest associations with microbial community structures and network topology in the UDS treatment, followed by the DS and D treatments (Figs. 2 and S6c).

3.4 Phosphate oxygen isotope signatures

As shown in Fig. 4b, all measured $\delta^{18}\text{O}_\text{p}$ values deviated from the equilibrium value ($E\delta^{18}\text{O}_\text{p} = 14.34\text{‰}$) previously reported for summer algae-bloom-affected Lake Taihu (Yuan et al., 2019). Higher $\delta^{18}\text{O}_\text{p}$ values were observed in sediment/soil samples, while the $\delta^{18}\text{O}_\text{p}$ value of SRP in the overlying water was $13.53 \pm 0.33\text{‰}$. Simulated dredging significantly reduced $\delta^{18}\text{O}_\text{p}$ values, from $16.09 \pm 0.13\text{‰}$ in the UDS treatment to $15.34 \pm 0.06\text{‰}$ and $15.65 \pm 0.15\text{‰}$ in the D and DS treatments, respectively. The

$\delta^{18}\text{O}_p$ values of surface sediment/soil at the incubation site and lakeshore were $16.09 \pm 0.13\text{‰}$ and $16.19 \pm 0.11\text{‰}$, respectively, while settling particles showed a value of $16.81 \pm 0.34\text{‰}$. The DS treatment showed significantly higher $\delta^{18}\text{O}_p$ values than the D treatment.

In the early stage of this study, simulated dredging reduced the sediment mobile P pool by 50.55% compared with the UDS treatment (Fig. 4a). After 180d under continuous particle deposition, benthic SRP efflux and apparent P recycling ratios increased during warm seasons (Fig. 3 and Table 1). Consistently, sediment mobile P pools in the DS treatment recovered to ~80% of those in the UDS treatment, whereas they remained low in the D treatment (Fig. 4a).

4. Discussion

4.1 Effects of sediment dredging on coupled cycles of carbon, phosphorus, and iron

In sediments, heterotrophic bacteria oxidize OM sequentially using electron acceptors in the order of O_2 , nitrate, Mn/Fe oxides, and sulfate (Burdige, 2006). Consequently, DO is preferentially consumed near the SWI, resulting in a shallow OPD (Fig. 1a). OPD is a sensitive parameter reflecting sedimentary OM reactivity (Glud, 2008). Shallower OPD indicating higher OM availability, consistent with the negative correlations between OPD and water-column Chl. a and sediment Org-P (Table S3 and Fig. 2), indicating that freshly deposited algal OM enhanced benthic oxygen consumption. The deeper OPD after simulated dredging reflects exposure of more refractory carbon at depth (Jørgensen et al., 2022). DO levels at the SWI were balanced by oxygen diffusion (water column DO, $r = 0.53$, $p < 0.05$) and sediment biochemical consumption (sediment LOI, $r = -0.54$, $p < 0.05$) (Fig. 1b). Improved DO levels were detected in the D treatment due to the restricted OM deposition (Sun et al., 2024; Lin et al., 2025). Overall, simulated dredging effectively removes OM-rich surface sediment, improving the sediment redox regime.

Sediment P cycling is strongly coupled to OM remineralization and redox-controlled Fe cycling (Li et al., 2018; Markovic et al., 2019). The subsurface Fe^{2+} peaks, except for the D treatment, favor Fe^{2+} ions either diffusing upward toward the overlying

water (recycling) or downward to depth (burial) through potential mineral sequestration under reducing conditions, possibly Fe-S interactions, as reflected by the convex curvature of the profiles (Lin et al., 2026). The correlations between porewater SRP and sediment LOI and Org-P indicate that porewater SRP release was driven partly by OM remineralization (Li et al., 2017b), whereas its relationship with Fe-P and porewater Fe^{2+} also suggests an important contribution from Fe-P coupling through oxidative sorption and reductive desorption processes (Tammeorg et al., 2020). Consequently, in surface sediment where intruding O_2 was available, Fe^{2+} could be oxidized to Fe(III) oxides, which efficiently scavenge SRP from ambient porewater (Weber et al., 2006; Ajmal et al., 2018). In addition, the availability of $\text{NO}_3^- \text{N}$ may further support nitrate-related anaerobic Fe oxidation, although this process was not quantified in this study (Roden and Wetzel, 2002; Sun et al., 2023, 2025). This interpretation is consistent with the negligible SRP and DGT-labile P concentrations observed within Fe^{2+} -depleted scales, except at 180d and 240d (Fig. 3).

The strong seasonal variability creates two contrasting P recycling regimes: during cold, well-oxygenated seasons (e.g., 60d and 360d), porewater SRP was efficiently sequestered by oxidized Fe(III) minerals, consistent with the negligible porewater SRP and Fe^{2+} concentrations (Fig. 3a and b) and temporary SRP influx observed during these periods (Table 1) (Li et al., 2018). These behaviors continuously scavenge SRP as sediment P, as supported by the low K_d coefficients in DIFS model (Table S4). In contrast, during warm seasons with low-oxygen availability (e.g., 180d and 240d), porewater SRP was recycled not only from organism decomposition (supported by substantial Org-P introduced by deposited SP (Figs. 4a and S2)), promoting early diagenesis (Monod kinetics model) (Westrich and Berner, 1984), and alkaline phosphatase activity (APA, an enzyme used for organic P compounds hydrolysis, Fig. S3) (Li et al., 2017a), but also from enhanced Fe-P reductive dissolution due to the redox microenvironment at the SWI, as suggested by the synchronously increased Fe^{2+} and SRP effluxes (Fig. 5b) (Li et al., 2017b; Wen et al., 2020). OM decomposition was the primary contributor as evidenced by similar porewater SRP concentrations between

UDS and DS treatments at 240d but apparent porewater Fe^{2+} differences between the three treatments because simulated dredging effectively removes Fe(III) oxides while resupply from SP needs time. Consequently, the sustained porewater SRP and Fe^{2+} profiles under SP influence indicate that particle deposition can rapidly re-establish strong P and Fe redox reactions and recycling even after simulated-dredging treatment (Kang et al., 2023).

The D treatment without SP deposition maintained consistently low porewater SRP and Fe^{2+} concentrations, weak gradients and resupply capacity (R ratio) throughout the study (Fig. 3 and Table S4). This pattern suggests that P remobilization was effectively suppressed by improving the redox environment and limiting external OM inputs as suggested by the stable Org-P content compared with the DS treatment (Fig. 4a). Together, our results demonstrate that the effectiveness of dredging in controlling sediment P remobilization is strongly contingent on post-dredging particle deposition, which can rapidly re-intensify OM mineralization, Fe redox reactions and associated P release.

4.2 SP inputs sustain elevated P efflux (sublinear SOU-SRP response) after dredging

Chemical fluxes across the SWI directly link sediment biogeochemistry and water column nutrient availability, thereby deciding the role of the sediment (source/sink) in aquatic ecosystems (Liu et al., 2016, 2019; Sun et al., 2025). The consistently larger SRP incubation fluxes than diffusive fluxes, together with their positive correlation, suggest that benthic faunal activity could enhance solute exchanges at the SWI, with possible additional contributions from chamber mixing, non-steady state gradients, and boundary-layer effects during incubation (Fig. 5a) (Burdige, 2006; Wen et al., 2020). The correlations between SRP fluxes, porewater SRP and Fe^{2+} , Fe^{2+} diffusive flux, sediment mobile P pools, and O_2 diffusive flux indicate that sediment SRP release was jointly controlled by redox-regulated Fe-P cycling, which enhanced synchronized SRP and Fe^{2+} release under low-oxygen conditions, and sediment OM remineralization processes (Hermans et al., 2021). Notably, source-sink transition was observed in all

three treatments during the cold-to-warm season shift, consistent with the two distinct P-cycling scenarios. In cold seasons, the overlying water was highly oxygenated and oxygen penetrated deep into the sediment, SRP diffusive effluxes were linearly regulated by SOU, and the slope was generally lower than the Redfield ratio, suggesting a fixed portion of SRP from OM decomposition was retained by oxidized Fe/Mn oxides in the surface oxic layer (Fig. 5a). This retention is supported by relatively low apparent P recycling ratios (< 1) and even negative SRP fluxes (sediment acting as P sink), as well as by generally increased BD-P content in DS and D treatment during the cold season (Fig. 4a). In contrast, this reserved P pool was substantially released in warm seasons as indicated by drastically increased P effluxes and apparent P recycling ratios (> 1), which suggests that SRP efflux was driven by both OM degradation and the remobilization of the previously retained P pool.

A nonlinear response of sediment SRP efflux to OM remineralization can amplify sediments biogeochemical feedback to the water column, indicating that sediment P release was promoted by OM diagenetic processes and modulated by other secondary effects. In this study, we reported a phenomenon that SRP efflux increased with SOU in a sublinear power-law form (Fig. 5a, $F_{SRP}=0.542*SOU^{0.410}$, exponent = 0.410, $r^2 = 0.930$) in the UDS and DS treatments during the warm seasons, which aligns with other observations in eutrophic water ecosystems (Sun et al., 2025; Lin et al., 2026). Despite the diminishing marginal increase, the observed relationship remained above the C:P ratio-based remineralization line, suggesting that SRP efflux was not driven by organic matter remineralization alone. Additional redox-mediated P remobilization, associated with tightly coupled sediment P, C, Fe, and S cycling under low-oxygen conditions, likely contributed to the elevated P release. During warm seasons, enhanced SP deposition increased surface sediment OM availability (increased LOI) and different P fractions relative to the D treatment (Fig. 4a and Fig. S2) (Li et al., 2018; Hermans et al., 2021; Wang et al., 2026). Therefore, enhanced microbial metabolism processes, as supported by higher O_2 influx, elevated microbial total and APA activity (Fig. S3), would directly increase SRP generation from OM mineralization while simultaneously

reducing the redox microenvironment at the SWI. The resulting constraint of the oxic surface layer reduced short-term P adsorption capacity, as suggested by the decrease in $Q_{\max}$ values in the DS treatment (Table S5 and Fig. S4). Meanwhile, enhanced anaerobic respiration further promoted SRP remobilization: reductive dissolution of ferric iron oxyhydroxides directly liberates Fe^{2+} and SRP back to the ambient porewater, consistent with the positive correlations between SRP and Fe^{2+} diffusive fluxes in UDS and DS treatments (Fig. 5b); in deeper sediment, the decrease in porewater Fe^{2+} may indicate removal of reactive Fe through mineral sequestration, potentially through Fe-S interactions, which could impede the regeneration of P-binding Fe oxides in the surface sediment and weaken long-term sediment P retention capacity. In addition, reduced NO_3^- -N availability in the overlying water during warm seasons (Table S3) may suppress potential nitrate-linked anaerobic Fe^{2+} oxidation, thereby promoting SRP efflux (Roden, 2012; Zhong et al., 2021). In contrast, similar warm-season amplifying phenomena were not evident in the D treatment in our study. Simulated-dredging removed OM- and reactive-Fe-enriched surface sediment, and the restricted SP deposition cut off the supply of labile OM and Fe, thereby weakening the coupling between OM remineralization, redox microenvironment and Fe-mediated P remobilization (Lüring and Faassen, 2012; Wu et al., 2025). This decoupling is supported by negligible chemical fluxes across the SWI, and also explains that dredging controlled the P recycling ratio after one-year of incubation (Table 1), demonstrating sediment dredging could manage internal loading in the long-term.

Modelling and evaluating the contribution of sediment internal P release is critical in models that predict water column P dynamics, although most such models parameterize internal P loading and flux using fixed sediment C:N:P ratios or empirical linear relationships, e.g., SRP efflux with sediment SOU or POC sedimentation rate (Fennel et al., 2022; Zhang et al., 2026). Our study instead provides two divergent, seasonal scenarios: effective sediment P sequestration in cold, oxygenated periods via oxic surface layer retention, but a warm-season regime in which benthic SRP efflux follows a sublinear relation with OM mineralization in UDS and DS treatments

contributed by OM aerobic respiration and other secondary effects as discussed above. Accordingly, eutrophic ecosystems with continuous external inputs may experience a phosphate warm pulse from sediment that will introduce systemic bias if models assume a constant P recycling efficiency; this simplification would tend to underestimate sediment P contribution in warm seasons but overestimate it in cold seasons. Additionally, SP inputs sustain this phenomenon by providing fresh OM and exacerbating redox conditions at the SWI, suggesting that dredging should be combined with other remediation (e.g., external pollutant management, macrophyte restoration, or in-situ capping) to suppress efflux pulses and maintain management of internal loads (Yin et al., 2021; Sun et al., 2023).

4.3 Effects of dredging and SP inputs on microbial community structures and co-occurrence patterns

Sediment diagenesis, particularly OM mineralization and redox-sensitive P-Fe-S cycling, is fueled by microbial metabolism, therefore changes in microbial community structure will further influence P mobilization following dredging and SP inputs (Zhu et al., 2021; Li et al., 2025). In this study, microbial richness in the DS treatment was significantly lower than that in the UDS treatment, whereas the evenness and Shannon-wiener index in the D were higher than those in the UDS and DS treatment (Fig. 6b). There were also significant ($n = 72$, pseudo- $F = 6.06$, $R^2 = 0.21$, $p < 0.05$) differences in microbial community structures, with the highest and lowest community dissimilarity measured in the UDS and D group (Fig. 6c and d). This pattern indicates that D treatment reduced microbial richness but produced a homogeneous community as dredging removed the nutrient-rich layer and created a stable geochemical environment (Li et al., 2026). Sediment microbial dynamics were effectively regulated by simulated-dredging treatment and SP inputs, with similar seasonal pattern of richness between UDS and SP groups but a distinct temporal lag in DS group, reflecting that dredging imposed a disturbance on sediment microbial communities but SP acted as a carrier of allochthonous microorganisms and substrate for sediment microbial recovery (Zhang et al., 2016).

The network topological features suggest a simplified network structure after simulated-dredging treatment, whereas SP inputs increase network connectivity (Fig. 7a and b; Table S7). This can be interpreted as dredging removed the surface sediment where intense redox reactions and complex geochemical gradients typically occur, thereby simplifying microbial interactions and restructuring functional modules (Chen et al., 2025). In contrast, settled particles favored the formation of specialized functional modules and species interactions. Consistently, the higher network robustness in the D treatment but lowest in the UDS treatment (Fig. 6c), suggests that simulated dredging stabilized the microbial network, whereas SP inputs diminished it. In addition, the PSB community composition also reflected responses to dredging and particle deposition. The dominance of *Acinetobacter* in both the UDS and DS treatments underscore its structural reliance on SP inputs for spatial colonization (Hu et al., 2022), and the enrichment of *Burkholderia* in DS indicated its stress-tolerant capability to rapidly adapt to physical disturbances and exploit newly available resources (Ramírez-Saad et al., 2000). In contrast, a high proportion of *Flavobacterium* was detected in SP but was low in DS and D groups, corroborating its known ability for sedimentary biopolymer degradation under high OM loading (Bissett, 2008). The PCoA analysis confirms the significant effect of sediment dredging on PSB community composition that was diminished by settling particles (Fig. S5b). Furthermore, the RDA result suggests that PSB closely interacts with ambient environments (Fig. S5c), for instance, OPD is significantly correlated with the abundances of *Thiobacillus*, *Pseudomonas* and *Dechloromonas*, demonstrating that the micro-scale redox boundary governs the coupled Fe-S-P solubilization via chemoautotrophic and facultative heterotrophic/hydrogenotrophic metabolisms in the bed sediments (Duffner et al., 2021; Cheng et al., 2024).

Microbial activities are regulated by the ambient environment and, in turn, reshape sediment geochemistry via diagenetic processes (Burdige, 2006). The dominance of *Chloroflexi-Chloroflexi* correlations and keystones in the UDS treatment may reflect a potential role of *Chloroflexi* in sediment anaerobic OM respiration. *Actinobacteria*,

which dominated in the DS treatment, are known for their ability to degrade complex OM using diverse extracellular enzymes, and their metabolism is enhanced by external nutrient inputs (Fan et al., 2025). Consequently, settling particles in the DS treatment introduced substrates that favored the proliferation of *Actinobacteria* and promoted mineralization of externally derived OM. In contrast, *Firmicutes-Firmicutes* correlations dominated in the D treatment because dredging and restricted SP inputs created an oligotrophic environment where *Firmicutes* may rely on sporulation to withstand stress (Cutting and Ricca, 2014).

The greater number and functional diversity of keystone taxa in the UDS treatment, including taxa associated with anaerobic fermentation, nitrification, and sulfate reduction, are consistent with more diverse sediment biogeochemical processes in undredged sediments (Table S8) (Prosser et al., 2014; McIlroy et al., 2017). In comparison, the dominance of taxa associated with sulfur-containing OM degradation in the DS treatment and dissimilatory iron reduction in the D treatment suggests a restructuring of potential microbial functions following dredging and SP deposition (Zhu et al., 2023; Kulichevskaya et al., 2025). The progressively weaker association between environmental variables and microbial community/network properties from UDS to DS and D further suggests that simulated dredging increased network robustness to ambient environmental variability (Fig. 2 and S6c). This reduced environmental sensitivity may reflect environmental selection for disturbance-resistant taxa with broader ecological adaptability (Wan et al., 2021).

3.4 Implications for lake internal loading management: sediment P sources identified by phosphate oxygen isotope

In this study, we applied phosphate oxygen isotopes ($\delta^{18}\text{O}_\text{P}$) to trace potential P sources in a dredged-sediment field system. Identifying the sources of sediment P loading is important for making management decisions in remediated eutrophication ecosystems. The deviation of all measured $\delta^{18}\text{O}_\text{P}$ values from the equilibrium value ($\text{E}\delta^{18}\text{O}_\text{P} = 14.34\text{‰}$) supports the applicability of the $\delta^{18}\text{O}_\text{P}$ technique as a qualitative source-tracing tool in this system (Fig. 4b). Higher $\delta^{18}\text{O}_\text{P}$ values were observed in

sediment/soil samples, while the $\delta^{18}\text{O}_\text{P}$ value of SRP in the overlying water (13.53±0.33‰) was close to equilibrium, suggesting rapid biomass-mediated P turnover in the water column and sedimentary P cycling in benthic sediment (Li et al., 2017a; Yuan et al., 2019). The decrease in sediment $\delta^{18}\text{O}_\text{P}$ values following simulated dredging likely reflects P cycling whereby the released SRP from enzyme-catalyzed organic matter hydrolysis subsequently co-precipitated as authigenic P minerals (e.g., Ca-P) at depth during burial (Jin et al., 2026). Under a practical dredging scenario, the newly exposed SWI is sensitive to particle deposition. Although overlapping $\delta^{18}\text{O}_\text{P}$ ranges among potential endmembers prevent quantitative distinction of their relative contributions, the significantly higher $\delta^{18}\text{O}_\text{P}$ values measured in the DS compared with the D treatment indicate that heavier endmembers, including surface sediment/soil at the incubation site and lakeshore, and settling particles, collectively contributed to the sediment P pool. Given the limited sampling coverage, $\delta^{18}\text{O}_\text{P}$ was used here to constrain the likely contribution of local sediment inputs to post-dredging P recovery, rather than to fully resolve all potential external sources. This pattern can be attributed to strong hydrodynamics in shallow lakes, which promote frequent resuspension of loosely local surface sediments and enhance their contribution to the water column and subsequent redeposition. In contrast, settled particles with higher $\delta^{18}\text{O}_\text{P}$ fingerprint derived from nonlocal sources appear to contribute less due to their relatively smaller abundance.

Dredging can be effective in controlling sediment internal nutrient loading, but its limited effective lifespan constrains the application of this expensive geoengineering approach (Wen et al., 2020; Yin et al., 2021). Although simulated dredging initially reduced the mobile P pool (Fig. 4a), however, P recycling was effectively re-established after 180d under continuous particle deposition, with benthic SRP efflux pulses and enhanced recycling efficiency during warm seasons (Fig. 3 and Table 1). Consistently, sediment mobile P pools in the DS recovered to ~80% of those in the UDS treatment, whereas they remained low in the D treatment (Fig. 4a). Contrary to the conventional view, our $\delta^{18}\text{O}_\text{P}$ fingerprint suggests that post-dredging P recovery was associated with local sediment-derived P inputs under strong hydrodynamic disturbance, reflecting

intensive intralake P recycling. Whilst sediment dredging is an essential step to interrupt intralake P recycling by removing contaminated sediments and thereby effectively suppressing SRP flux and recycling for the long term, its performance is strongly constrained by post-dredging particle dynamics. Therefore, dredging should be integrated with complementary management approaches, e.g., in-situ P immobilization capping and bottom-water aeration, to suppress warm-season P release (Yin et al., 2020; Testa et al., 2025). Notably, the simulated-dredging experiment does not fully reproduce full-scale dredging practices, where residual contaminated sediments, sediment disturbance and resuspension may coexist. In addition, although the mesh effectively excludes SP inputs in the D treatment, it may also have modified benthic faunal activity, hydrodynamics, and diffusive boundary development near the SWI. These factors may strongly influence sediment P mobility and release, and thus the apparent performance of sediment dredging (Chen et al., 2026). Nevertheless, these findings have direct implications for dredging practice, highlighting the importance of low-residual approaches (e.g., cutter-suction) to minimize post-dredging particle and OM inputs and thereby prolong the suppression of sediment P recycling. Future research should focus on long-term performance monitoring in full-scale dredging projects, as well as integration with complementary geoengineering technologies.

4. Conclusion

(1) Sediment dredging effectively suppressed sediment P remobilization by removing OM-rich surface sediment and improving redox conditions at the SWI, but its effectiveness was constrained by post-dredging settling particle deposition.

(2) Settling particles rapidly reintroduced reactive OM and iron, reactivating early diagenesis and coupled C-Fe-P recycling, particularly under warm-season reducing conditions; a sublinear SRP efflux vs. sediment oxygen uptake demonstrated that sediment P release was governed not only by OM remineralization but also by secondary redox-mediated feedbacks.

(3) Dredging simplified microbial network structure and enhanced network stability, whereas settling particle inputs weakened this effect and reinforced coupled

P–Fe–S biogeochemical cycling. In addition, $\delta^{18}\text{O}_p$ fingerprints indicated that post-dredging sediment recovery was associated with the redeposition of local sediment-derived particles.

(4) Sediment dredging is therefore an important step for interrupting intralake P recycling, but sustained control of internal P loading requires complementary management measures that limit sediment resuspension and particle redeposition. Future remediation strategies should integrate dredging with complementary geoengineering approaches to prolong treatment effectiveness.

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Table 1. Intact-core incubation fluxes of SRP and diffusive fluxes of SRP, Fe²⁺, and O₂ across the SWI, and apparent P recycling ratio in surface sediments of three treatments throughout the incubation period.

Treatments		Intact-core incubation fluxes (mg·m ⁻² ·d ⁻¹)	Diffusive fluxes (mg·m ⁻² ·d ⁻¹)			Apparent P recycling ratio
		SRP	SRP	Fe ²⁺	O ₂	
0d	UDS	-0.140±0.119	-0.096	0.435	-63.778	N/A
	D	-0.259±0.259	0.128	0.198	-18.391	0.96
60d	UDS	-0.931±0.091	-0.017	-0.062	-18.908	N/A
	DS	-2.326±0.053	-0.156	0.004	-11.830	N/A
	D	-1.520±0.120	-0.042	0.013	-4.338	N/A
120d	UDS	3.566±1.444	0.027	5.715	-31.623	0.12
	DS	1.383±0.555	0.168	0.419	-80.460	0.29
	D	1.472±0.162	0.068	0.010	-10.217	0.92
180d	UDS	4.890±0.898	4.669	17.222	-201.633	3.20
	DS	3.969±1.069	1.997	22.475	-31.939	8.63
	D	1.162±0.291	0.181	3.384	-7.801	3.20
240d	UDS	5.309±0.166	3.347	14.416	-49.609	9.31
	DS	5.486±1.937	2.332	1.731	-43.948	7.32
	D	0.446±0.186	0.385	0.284	-58.064	0.91
300d	UDS	2.788±1.879	0.357	1.331	-140.841	0.35
	DS	2.286±0.693	0.854	2.371	-4.588	25.69
	D	-0.679±0.367	-0.196	0.149	-6.634	N/A
360d	UDS	-0.603±0.633	0.120	0.750	-22.690	0.73
	DS	-1.530±0.036	0.039	0.438	-57.081	0.09
	D	-2.409±0.486	0.00005	0.146	-7.320	0.00

Note: Positive fluxes represent flux out from sediment to overlying water, whereas negative fluxes represent flux from the overlying water into the sediment. The apparent P recycling ratio was calculated as the ratio of SRP diffusive effluxes over SRP production from organic matter remineralization ([P efflux]:[P production]), and a higher ratio suggests greater P recycling back to the overlying water. P production is calculated using SOU under Redfield ratio (138O₂: 106C: 16N: 1P) assumption. A sensitivity analysis using alternative O₂:P ratios is provided in the Supplementary Materials (Table S9).

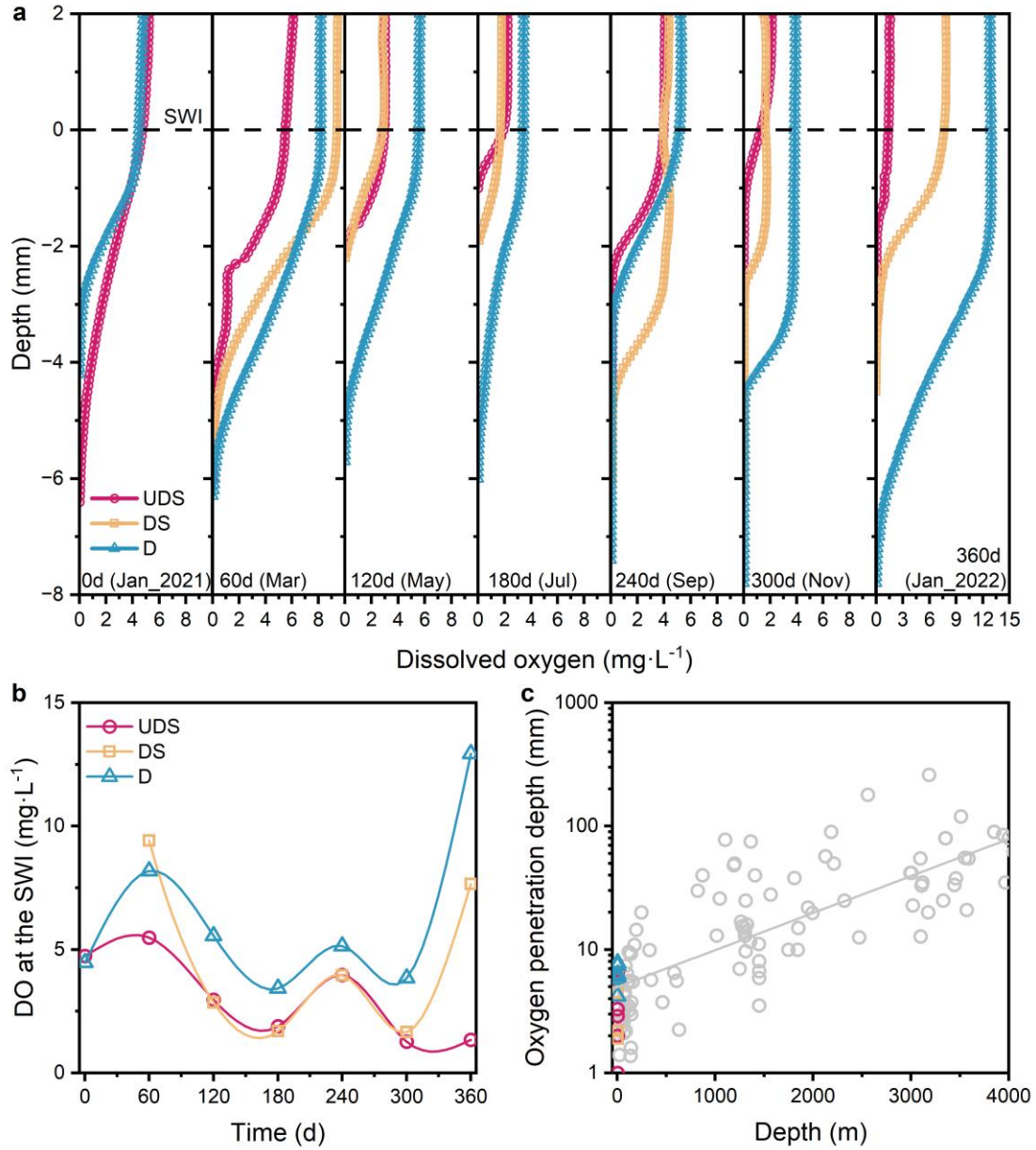


Figure 1. (a) Micro-profiles of dissolved oxygen at the SWI. The dashed horizontal line at the zero depth indicates SWI; (b) variation of DO concentration at the SWI of three treatments during the incubation period; and (c) sediment oxygen penetration depth vs. water depth. Global data and the linear regression are adopted from Jørgensen et al., (2022).

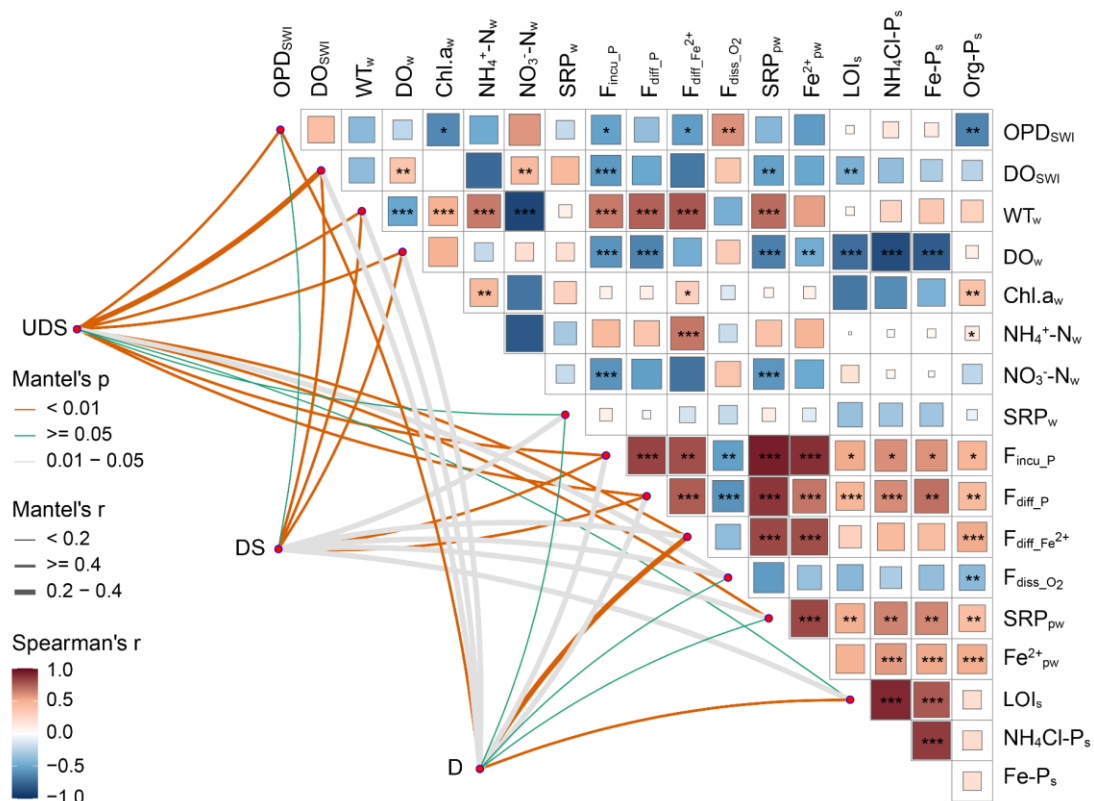


Figure 2. Correlations between environmental factors and microbial communities across UDS, DS, and D treatments.

Note: SWI: sediment-water interface, W: overlying water, incu/diff_i: incubated/diffusive flux of interests i, pw: porewater, and S: sediment.

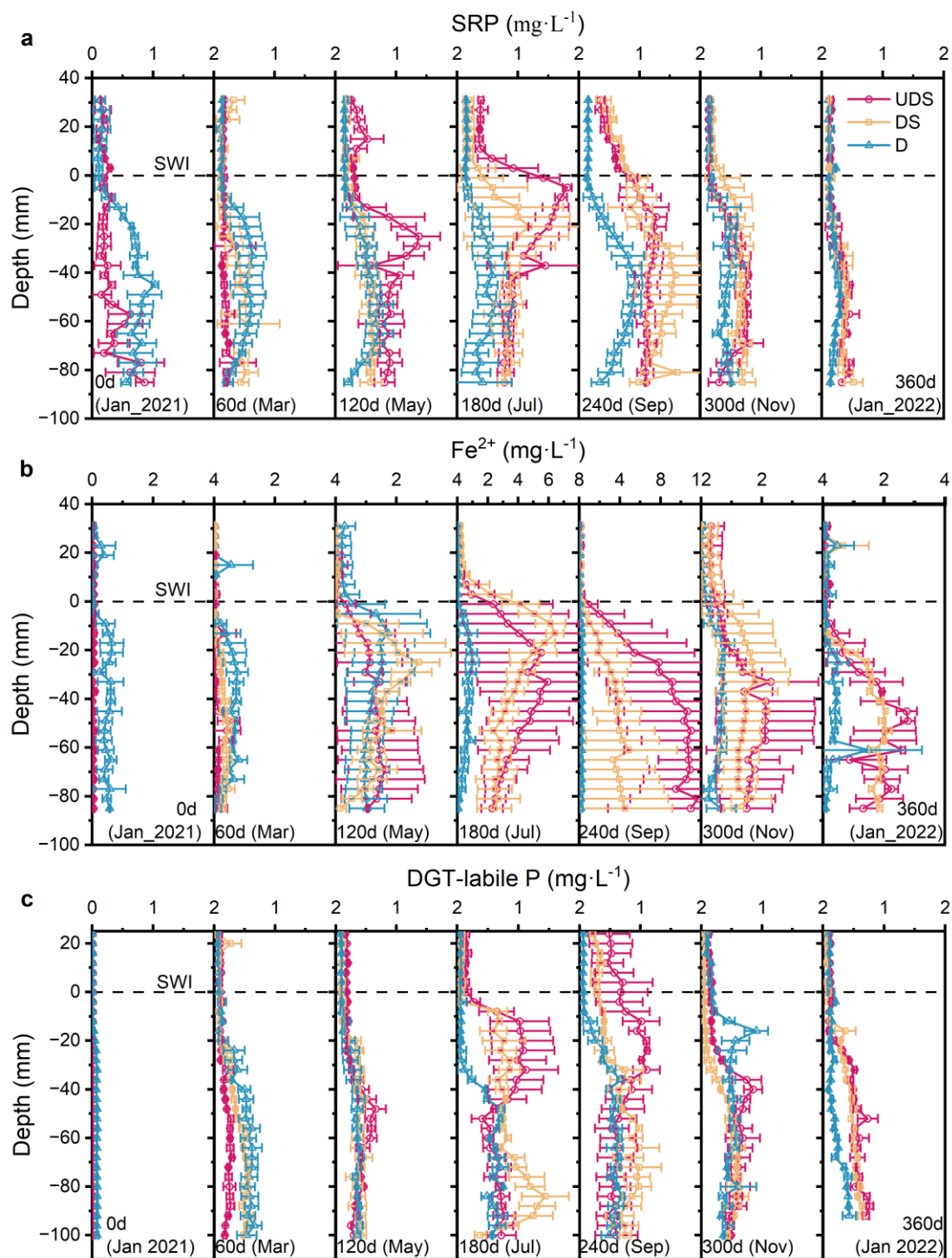


Figure 3. High-resolution porewater (a) SRP, (b) Fe^{2+} , and (c) DGT-labile P profiles at the SWI of UDS, DS, and D treatments throughout the incubation period.

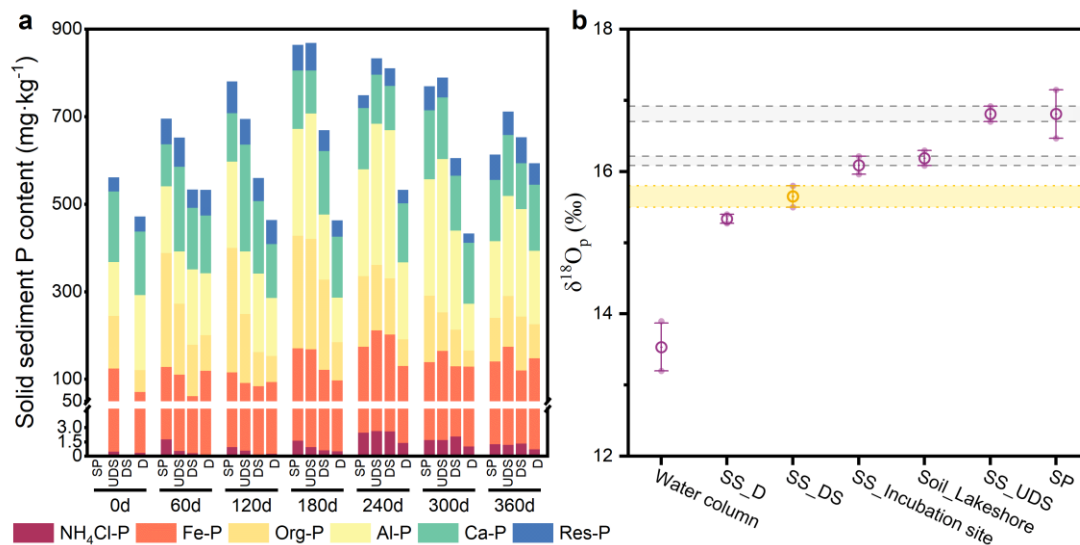


Figure 4. (a) Solid phase of sediment P fractions in surface sediment (SS, 0-2cm) of three treatments and SP throughout the incubation period; and (b) oxygen isotopic composition of dissolved phosphate in different potential P sources.

Note: In panel (b), small purple points represent individual measurements, circles with error bars represent mean ± SE, and grey dashed lines indicate the overlapping δ¹⁸O_p range among potential endmembers.

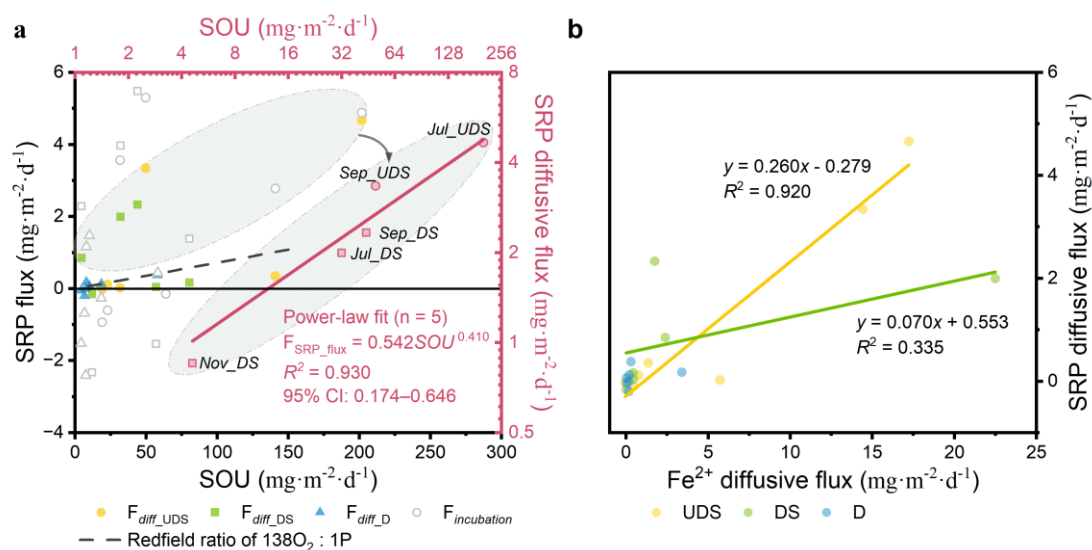


Figure 5. (a) Sediment oxygen uptake (SOU, shown as positive O_2 consumption) rates vs. sediment SRP flux, and SOU vs. SRP diffusive effluxes on a log-log axis (nonlinear relationships) during summer seasons, grey symbols are indicated of intact-core incubation fluxes. The sample size and 95% confidence interval are shown in the panel, and residual diagnostics (Fig. S7) and influential-point analysis (Table. S10) are available in the SI; and (b) linear regression of SRP diffusive flux vs. Fe^{2+} diffusive flux in three treatments.

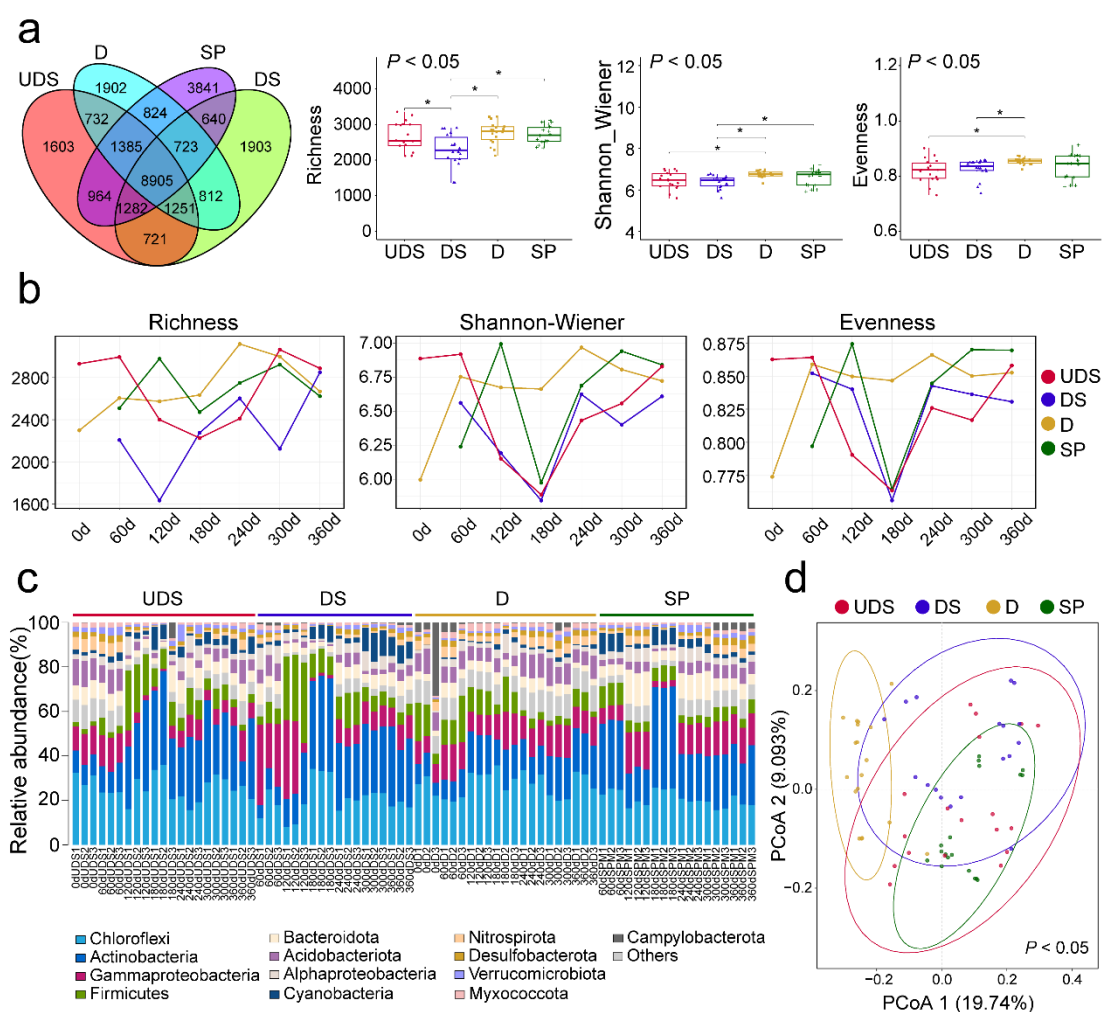


Figure 6. Microbial diversity and community composition across three treatments. (a) Shared and unique OTUs, and alpha-diversity indices. (b) Temporal variation in alpha diversity. (c) Relative abundance of dominant microbial taxa (>5%). (d) PCoA of microbial communities. Significant differences are indicated at $p < 0.05$.

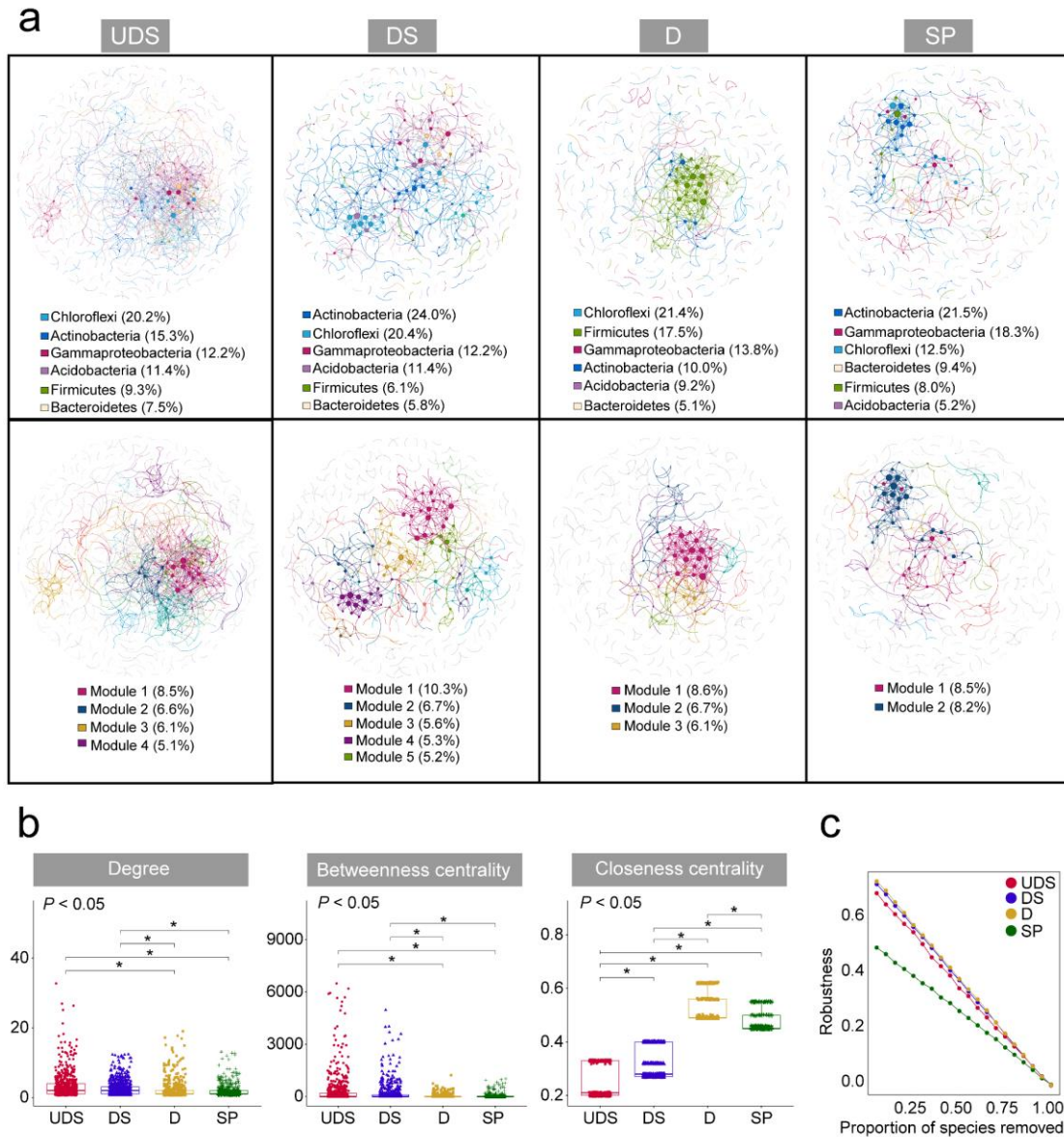


Figure 7. The structure, topological parameters and robustness of networks. (a) The networks among microbial communities across different groups. (b) Node-level topological parameters. (c) Microbial robustness vs. the proportion of removed nodes. Note: Major (> 5%) taxonomic groups and modules of network were shown in figure a.

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