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Global meta-analysis deciphering ecological restoration performance of dredging: Divergent variabilities of pollutants and hydrobiontes

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A B S T R A C T

Global "Sustainable Development Goals" propose ambitious targets to protect water resource and provide clean water, whereas comprehensive understanding of restoration performance and ecological mechanisms are lacking for dredging adopted for purifying polluted waterbodies and maintaining navigation channels. Here, we conducted a global meta-analysis to estimate ecological restoration consequence of dredging as pollution mitigation and navigation channel maintenance measures using a dataset compiled from 191 articles covering 696 studies and 84 environmental and ecological parameters (e.g., pollutants and hydrobiontes). We confirm that dredging shows negative influences on 77.50% pollutants in the BA model (before dredging vs. after dredging) and 84.21% pollutants in the CI model (control vs. impact) as well as on sediment nutrient fluxes. Additionally, 57.14% attributes (i.e., richness, diversity, biomass, and density) of hydrobiontes in the BA model and 89.47% attributes of hydrobiontes in the CI model responded negatively to dredging. As a result, 76.32% of the pollutants and 61.11% of the hydrobiont attributes responded uniformly to dredging in the BA and CI models. Our findings emphasize that dredging generally decreases pollutants and mitigates algal blooms, controlling phosphorus is easier than controlling nitrogen by dredging, and attributes (i.e., richness, diversity, and biomass) of hydrobiontes (i.e., zooplankton, phytoplankton, and zoobenthos) are density-dependent in dredging-disturbed environments. Our findings broaden our knowledge on ecological restoration performance of dredging as a mitigation measure in global aquatic ecosystems, and these findings might be helpful to use and optimize dredging to efficiently and sustainably purify polluted aquatic ecosystems.

1. Introduction

In the Anthropocene, global freshwater and seawater ecosystems are facing severe anthropogenic pollution by nutrients (e.g., nitrogen and phosphorus), heavy metals (e.g., mercury and cadmium), organic compounds (e.g., aromatic compounds and microplastics), and radioactive contaminants (e.g., nuclear wastewater containing radioactive strontium) (Hou et al., 2022; Jones et al., 2023). Excessive pollutants in waterbody jeopardize aquatic biodiversity and thus their ecosystem services such as drinking water supply and purification, and consequently lead to great ecological and health risks to the world (McCutcheon et al., 2021). Pollutants in aquatic environments can be

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buried in the sediments via physical adsorption and chemical precipitation, and pollutants can migrate from the sediment to the water column via microenvironmental changes and biogeochemical cycles (McGrath et al., 2013; Wan et al., 2020). Therefore, it is essential to prevent resolution and migration of pollutants from the sediment to the waterbody by applying appropriate ecological restoration measures.

Dredging, currently regarded as an important restoration tool by physical means, shows great potentials in purifying various water bodies by removing sediments and plants to modify microenvironments (Lürling and Faassen, 2012; Oldenborg and Steinman, 2019). Ecologists investigating ecological consequences of environmental disturbance (e.g., dredging and flooding) have developed sampling strategies to evaluate changes in studied sites, and the BACI (Before-After-Control-Impact) sampling designs are regarded as the most rigorous method (Underwood, 1994). As for dredging study, there are studies considering impact before dredging vs. after dredging (BA model, factor level after dredging compared to before dredging) and impact un-dredged vs. dredged (CI model, factor level in dredged sites compared to un-dredged sites). Understanding ecosystem consequences of dredging disturbances in the BA and CI models are important to estimate whether dredging should be adopted to purify polluted environments. Different responses of nutritional pollutants (Jing et al., 2013; Wan et al., 2021a), heavy metal pollutants (Fathollahzadeh et al., 2015), organic pollutants (Cutroneoet al., 2015), and organisms (Zhang et al., 2017) to dredging have been previously reported for BA and CI models in different aquatic ecosystems (e.g., lakes and rivers). Consequently, it is necessary to comprehensively and systematically understand how various pollutants and aquatic organisms respond to dredging in aquatic ecosystems worldwide.

Excessive pollution by nutrients (e.g., nitrogen and phosphorus) in aquatic ecosystems can cause water eutrophication and, therefore, trigger algal blooms (Severiano et al., 2023), and thus most dredging-related studies focus on estimating dredging effects to weaken water eutrophication and mitigation of algal blooms by monitoring changes in nutrients and chlorophyll-a (Chl-a) in water-sediment systems (Oldenborg and Steinman, 2019; Wan et al., 2021a). Dredging has been reported to mitigate algal blooms when applying both the BA (Kang et al., 2023) and CI (Lürling and Faassen, 2012) models. Dredging shows generally positive effects on improving water quality according to earlier case studies, e.g., decreases have been observed in water total nitrogen and total phosphorus in the Lake Dongting BA model (Li et al., 2020) and in the Pond De Ploeg CI model (Lürling and Faassen, 2012). Dredging also shows negative influences on decreasing water total nitrogen and total phosphorus in the Suyahu Reservoir BA model (Gao et al., 2019) and in the Pond Bouvigne CI model (Kang et al., 2023). Dredging decreases sediment total nitrogen and total phosphorus load in the Lake Chaohu in the BA model (Liu et al., 2019) and in the Lake Dongqian in the CI model (Jing et al., 2013), and dredging also increases sediment total nitrogen and total phosphorus (Wan et al., 2022). Nitrogen and phosphorus release rates from sediments can be used to estimate migration potentials of nutrients from sediment to the water column, and dredging have differently affected these release rates in earlier studies (Gu et al., 2016; Zhong et al., 2018). Yet, information is insufficient regarding response differences of nutritional pollutants to dredging between both BA and CI models in aquatic ecosystems worldwide as well as mechanisms underlying dredging-induced mitigation of algal blooms.

Aquatic organisms (e.g., zooplankton and phytoplankton) are important components of any aquatic food web (Weitere et al., 2018), and their richness and diversity drive aquatic ecosystem multifunctioning (Moi et al., 2021; Zhang et al., 2021). Earlier studies report that dredging shows different effects on hydrobiont density, biomass, richness, and diversity in divergent aquatic ecosystems (Zhang et al., 2017; Li et al., 2020; Wan et al., 2022). For example, dredging decreases density and diversity of both zooplankton and phytoplankton in the Sepetiba Bay in the BA model (Fernandes et al., 2023), and dredging decreases richness and density of zooplankton in the Krapiel River in the CI model (Szlauer-Łukaszewska and Zawal, 2014). Additionally, dredging increases biomass and density of zoobenthos in the North Sea coast in the BA model, but the opposite pattern is found with the CI model (Rehitha et al., 2017). Today, differences in hydrobiont responses to dredging between BA and CI models remain unknown in global aquatic ecosystems as well as how hydrobiontes maintain their richness and diversity in environments with severe dredging disturbances.

Dredging is often implemented in inland aquatic ecosystems (e.g., lakes, rivers, ponds, and reservoirs) and coastlines (e.g., sea coast and sea bay), and dredging undertakes ecological restoration purposes, including cleanup projects (e.g., decreasing pollutants and improving water quality) and navigation channel maintenance (e.g., embankment reconstruction, modifying water flow, and protecting biodiversity). In the past decades, there are important dredging events in global aquatic ecosystems (Fig. 1a), including Lake Taihu in China (Chen et al., 2021), Lake Muskegon in American (Oldenborg and Steinman, 2019), Mississippi River in North America (Moore et al., 2017), and Krapiel River in Poland (Zawal et al., 2016). To unveil the knowledge gaps mentioned above, we conducted a global meta-analysis to summarize basic findings and explain potential mechanisms involved in dredging studies with restoration purposes in global aquatic ecosystems (Fig. 1b). Here, we aimed to (i) evaluate responses of pollutants and hydrobiontes to dredging as an ecological restoration approach in globally different ecosystem when using BA and/or CI models, (ii) decipher abiotic and biotic factors affecting the degree of algal blooms, and (iii) explain mechanisms of how hydrobiontes maintain a high diversity and biomass in dredging-disturbed aquatic environments. Considering dredging decreases nutrients and causes ecological drifts (Wan et al., 2021a), we hypothesize that most tested factors belonging to pollutants and hydrobiontes would respond uniformly (i.e., both positive or negative) to dredging in BA and CI models. Given population density and community diversity are positively connected to each other (Azizan et al., 2023), we hypothesize that there is a close link between density and α-diversity of hydrobiontes in BA and/or CI models.

2. Materials and methods

2.1. Data acquisition

To establish a comprehensive database to evaluate the ecological restoration performance of dredging, we collected experimental data from all relevant literature by searching keywords from the *Web of Science, Google Scholar*, and China National Knowledge Infrastructure. These keywords comprised the combination of “sediment * dredg”, “dredg * pollutant”, “dredg * nitro”, “dredg * phos”, “dredg * metal”, “dredg * algal bloom”, “dredg * chlorophyll”, “dredg * release rate”, and “dredg * organism”. Dredging-related articles without distinct ecological restoration aspects are excluded, such as dredging for aquatic products and sand with economic purposes, dredging for initial construction of ports, and dredging for opening ocean shipping routes. A PRISMA flow diagram was constructed to display step-by-step strategies for the selection of suitable references (Fig. S1).

The following criteria were used to enrich our database: (i) treatment group (i.e., after dredging and dredged region) and control group (i.e., before dredging and non-dredged region) were conducted in various aquatic ecosystems; (ii) treatment and control groups of outdoor dredging operations or indoor simulation experiments; (iii) paired treatment and control groups in a study; (iv) values and sample sizes in both treatment and control groups were directly reported or could be obtained by using software GetData Graph Digitizer (version 2.2.5, https://sourceforge.net/projects/getdata/) or Mean Variance Estimation website (https://www.math.hkbu.edu.hk/~tongt/papers/median2mean.html).

In total, we collected data from 696 studies on dredging reported, which are published in 191 papers. The separate BA and CI models

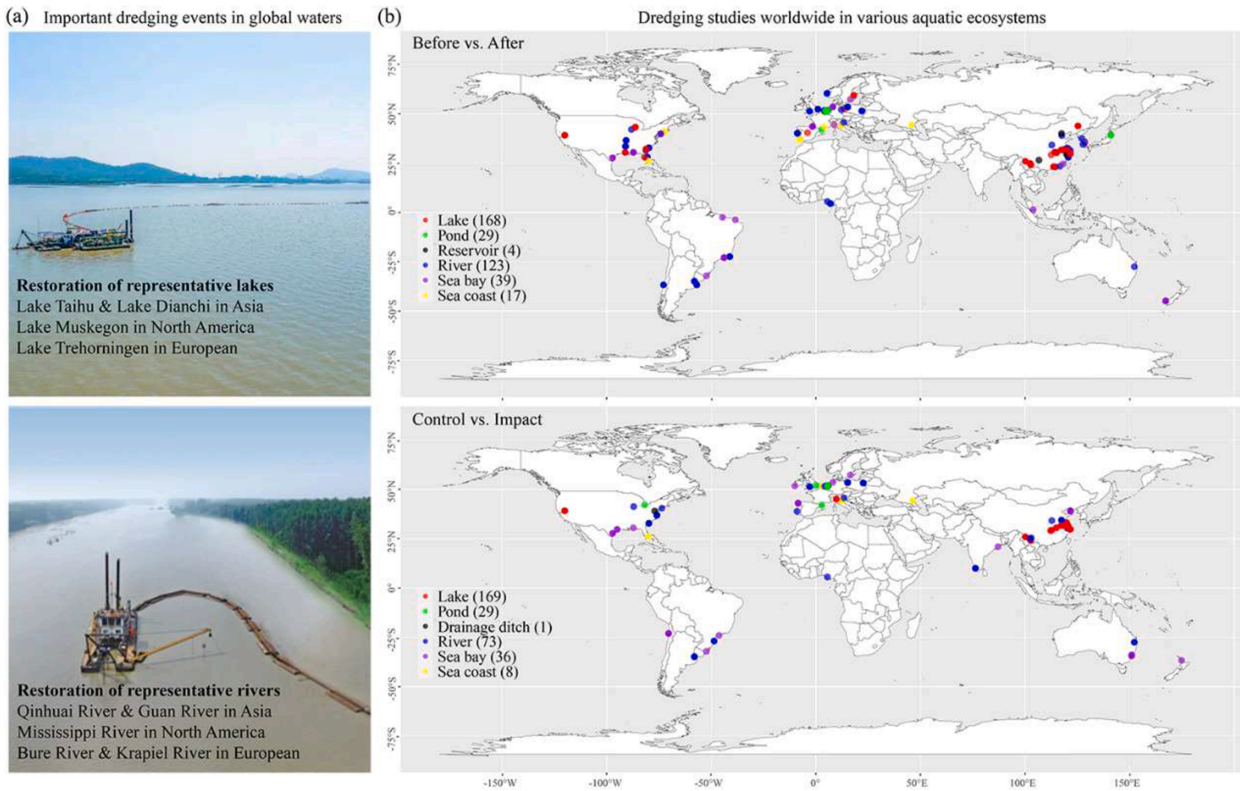

Fig. 1. Dredging events and dredging studies in global waters. (a) Ecological restoration of dredging events in global representative lakes and rivers. (b) Global distributions of studies included in this meta-analysis covering 380 studies in the BA model (before vs. after) and 316 studies in the CI model (control vs. impact). Numerical values in boxes denote numbers of studies in corresponding aquatic ecosystems receiving ecological restoration of dredging.

comprised 380 and 316 studies, respectively, which were separately from 129 and 103 articles. Aquatic ecosystems were classified as lake, river, pond, reservoir, sea bay, sea coast, and drainage ditch in the BACI model. Both BA (i.e., lake [44.21%], river [32.37%], pond [7.63%], reservoir [1.05%], sea bay [10.26%], and sea coast [4.47%]) and CI (i.e., lake [53.48%], river [23.10%], pond [9.18%], sea bay [11.39%], sea coast [2.53%], and drainage ditch [0.32%]) models comprised six aquatic ecosystems located in six continents around the world (i.e., Asia, Africa, European, Oceania, North America, and South America) (Fig. 1b).

Matching data were extracted and recoded, including water non-nutritional parameters (e.g., pH, dissolved oxygen, and Chl-a), nutritional pollutants (e.g., total nitrogen, total phosphorus, and organic matter), heavy metal pollutants (e.g., mercury, cadmium, and copper), organic pollutants (e.g., polyaromatic hydrocarbons and polychlorinated biphenyls), enzyme activity (e.g., β−1,4-N-acetyl-glucosaminidase and phosphatase), release rate (e.g., soluble reactive phosphorus and ammonia nitrogen), and hydrobiontes (e.g., zooplankton density). Additionally, recovery time after dredging and terrain properties (i.e., longitude and latitude) were also extracted from all scanned papers.

The publication time of studies in the BA model ranged from 1975 to 2023, whereas the CI model ranged from 1981 to 2023. The recovery time after dredging in the BA model ranged from 0.008 to 240 months, whereas the CI model ranged from 0.033 to 1224 months. We displayed the value range, mean value, unit, and sample number for each parameter mentioned above for the BA (Table S1) and CI (Table S2) models.

2.2. Meta-analysis

We used a meta-analysis to evaluate responses of abiotic (e.g., water general parameters and nutritional pollutants) and biotic (e.g., diversity and biomass of organisms) properties to dredging. To do so, we extracted the mean value, standard deviation (SD), and sample size (N) from the published studies. However, there were often missing values of SD and N of collected data. Subsequently, we completed the paired datasheet (i.e., mean value, SD, and N) for the BA (Table S3) and CI (Table S4) models by using reported standards (Hou et al., 2020).

(i) If standard error (SE) rather than SD was reported, SD was calculated as:

$$SD = SE \times \sqrt{N} \tag{1}$$

(ii) If neither SD nor SE was reported, we calculated the average coefficient of variance (CV) for our complete dataset, and thereafter SD was calculated as:

$$SD = \text{mean value} \times \text{average } CV \tag{2}$$

(iii) If sample size was not reported, we used the median sample size of our complete dataset to designate the missing N. The computed CV value and N for the BA (Table S3) and CI (Table S4) models are summarized in the supplementary information.

Effects of dredging on parameters (e.g., water general properties, nutritional pollutants, and hydrobiontes) were evaluated by calculating natural logarithm-transformed response ratio Ln(RR). The Ln(RR) was calculated as follows:

$$Ln(RR) = \ln \frac{\overline{X}_t}{\overline{X}_c} \tag{3}$$

where $\overline{X}_t$ is the mean value of the treatment group (i.e., after dredging and dredged region), and $\overline{X}_c$ is the corresponding mean value from the control group (i.e., before dredging and un-dredged region).

The weighted mean response ratio of Ln(RR$_+$) was calculated as follows:

$$Ln(RR_+) = \frac{\sum_{j=1}^{m} w_j^* \times \mathrm{Ln}(RR_j)}{\sum_{j=1}^{m} w_j^*} \tag{4}$$

where m is the number of experiments in the group, and w_j^* is the weighting factor of the jth experiment in the group. The w_j^* was computed as follows:

$$w_j^* = \frac{1}{v_j^*} \tag{5}$$

where v_j^* is the variance of study (j) in the group. The v_j^* was computed as follows:

$$v_j^* = \frac{S_t^2}{n_t X_t^2} + \frac{S_c^2}{n_c X_c^2} + \tau^2 \tag{6}$$

where n_t and n_c separately represent the sample sizes for the treatment group and the control group of the study (j), and S_t and S_c separately denote the standard deviations for the treatment group and the control group of the study (j). τ^2 is the between-studies variance.

The standard error of Ln(RR$_+$) was computed as:

$$S(Ln(RR+)) = \sqrt{\frac{1}{\sum_{j=1}^{m} w_j^*}} \tag{7}$$

The 95% confidence interval (CI) for the Ln(RR$_+$) was calculated as follows:

$$95\%CI = Ln(RR_+) + 1.96S(Ln(RR_+)) \tag{8}$$

If the 95%CI did not overlap with the zero value, it denotes significance. The percentage change was calculated as follows:

$$Effect\ size(\%) = \left(e^{(LnRR_+)} - 1\right) \times 100\% \tag{9}$$

We checked the quality of our meta-analysis based on the checklist of a previous protocol (Koricheva and Gurevitch, 2014), and our meta-analysis fulfilled all quality criteria for a meta-analysis (Table S5). The random-effect model was adopted based on the results of the Homogeneity test (Q test) used for meta-analysis (Table S6). We created funnel plots to detect possible publication biases using the "funnel" function, and we estimated the symmetric or asymmetric distribution of the weighted Ln(RR) in the funnel plots (Table S7). We further estimated publication biases by using the trim-and-fill method based on the "trimfill" function, and original values of Ln(RR) vs. trim of Ln(RR) are summarized in the supplementary materials (Table S8). Finally, we investigated the linkages between the Ln(RR) and publication year

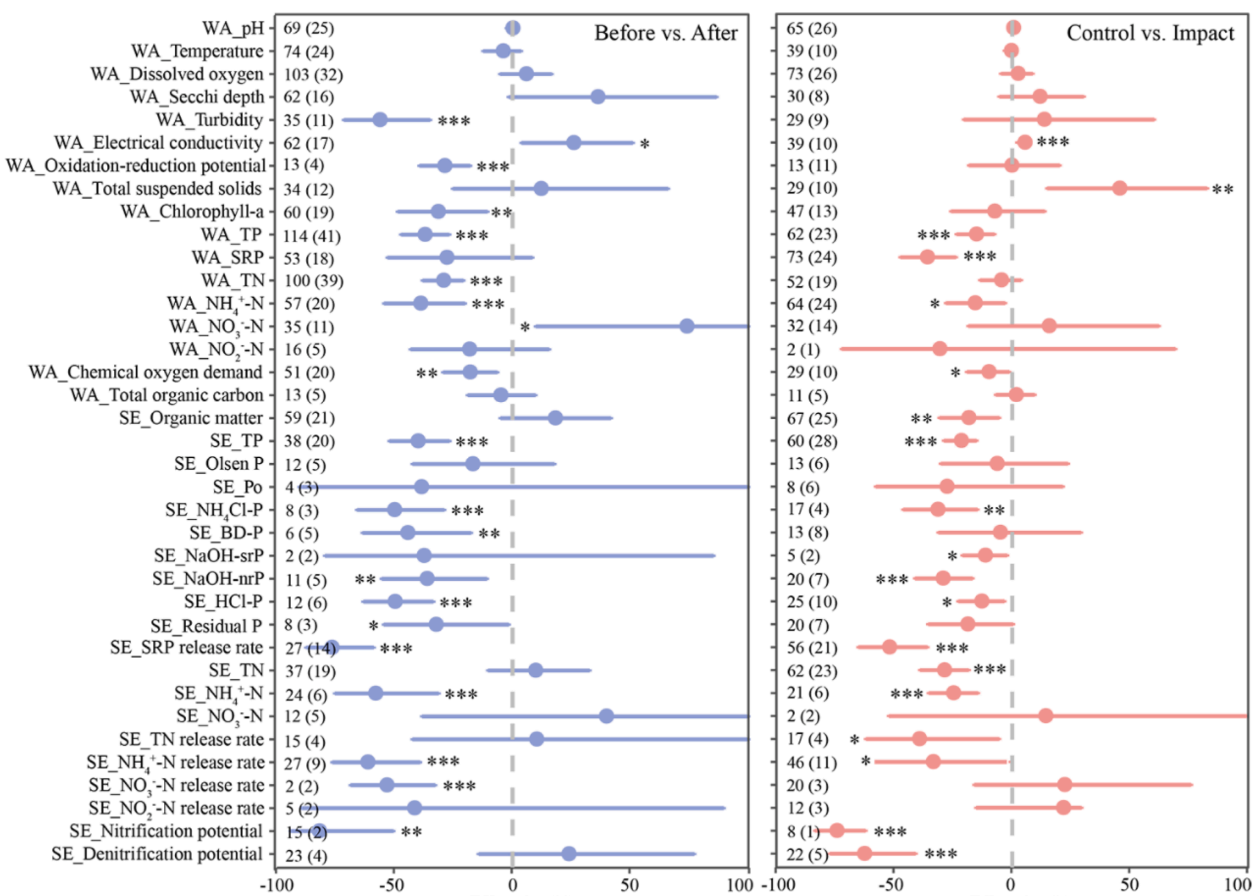

Fig. 2. Effects of dredging on water non-nutritional factors, water and sediment nutritional factors, and sediment nutrient release rate in the BA (before vs. after) and CI (control vs. impact) models. Values represent effect sizes ± 95% confidence intervals, and asterisks denote significance (*, $p < 0.05$; **, $p < 0.01$; ***, $p < 0.001$). Dashed lines represent lnRR = 0 and numbers within and outside the parentheses represent the number of articles and studies, respectively. Different values (i.e., mean values represented by blue and red dots) of "effect size (%)" for single parameter denote that there is difference between BA and CI models. Abbreviations: WA, water; SE, sediment; TP, total phosphorus; SRP, soluble reactive phosphorus; TN, total nitrogen; NH_4^+-N, ammonia; NO_3^--N, nitrate; NO_2^--N, nitrate; Po, organic phosphorus; NH_4Cl-P, NH_4Cl extractable phosphorus; BD-P, $NaHCO_3/Na_2S_2O_4$ extractable phosphorus; NaOH-srP, NaOH extractable reactive phosphorus; NaOH-nrP, NaOH no-reactive phosphorus; and HCl-P, HCl extractable phosphorus.

(Table S9) and the respective recovery time after dredging (Table S10).

The normal distribution of Ln(RR) was estimated using the Kolmogorov-Smirnov test (Table S11). Meta-regression was performed based on the least square method. The structural equation model was built by using IBM Amos 21. Statistical analyses mentioned above mainly adopted of the "ggplot2", "ggthemes", "glmulti", and "metafor" of R (version 4.2.1; https://www.r-project.org), and relevant R codes are summarized in the supplementary materials.

3. Results

3.1. Responses of algal blooms and pollutants to dredging

By estimating the percentage of effect size (Fig. 2), the Chl-a displayed a notable decrease in the BA model ($p < 0.01$), but a non-significant decrease in the CI model ($p > 0.05$). The pH and temperature exhibited slight changes in the BA and CI models ($p > 0.05$). There was also a significant increase ($p < 0.05$) in electrical conductivity and insignificant ($p > 0.05$) increases in dissolved oxygen and Secchi depth in both the BA and CI models, thereby the chemical oxygen demand was notably decreased in the BA model ($p < 0.01$) as well as the CI model ($p < 0.05$). Total suspended solids displayed an insignificant increase in the BA model ($p > 0.05$), but increased significantly in the CI model ($p < 0.01$), whereas the oxidation–reduction potential remarkably decreased in the BA model ($p < 0.01$), but insignificantly in the CI model ($p > 0.05$). These results - though partly different between the two models - indicate that dredging has the potential to mitigate algal blooms and change water microenvironments.

According to results of the percentage of effect size (Fig. 2, Fig. S2), pollutants in the water vs. the sediment responded largely differently to dredging, whereby results of both BA and CI models differed to some extent. In the water column, total phosphorus (TP), total nitrogen (TN), ammonia nitrogen (NH_4^+-N), copper, and iron, whereas in the sediment, TP, NH_4Cl extractable P (NH_4Cl-P), $NaHCO_3/Na_2S_2O_4$ extractable P (BD-P), NaOH no-reactive P (NaOH-nrP), HCl extractable P (HCl-P), residual P, NH_4^+-N, arsenic, chromium, copper, nickel, zinc, and polychlorinated biphenyls notably decreased in the BA model ($p < 0.05$; $p < 0.01$ or $p < 0.001$). In contrast, physicochemical properties of water including TP, soluble reactive P (SRP), NH_4^+-N, and arsenic and those in the sediment including organic matter, TP, NH_4Cl-P, NaOH extractable reactive P (NaOH-srP), NaOH-nrP, HCl-P, TN, NH_4^+-N, arsenic, cadmium, chromium, mercury, lead, and zinc significantly decreased in the CI model ($p < 0.05$; $p < 0.01$ or $p < 0.001$). There were also non-significant decreases in pollutants (Fig. 2, Fig. S2; $p > 0.05$), including pollutants in water (i.e., SRP, nitrite [NO_2^--N], total organic carbon, chromium, and lead) and sediment (i.e., Olsen P, organic P [Po], NaOH-srP, cadmium, iron, mercury, lead, polyaromatic hydrocarbons) in the BA model as well as pollutants in water (i.e., TN, NO_2^--N, cadmium, copper, iron, mercury, nickel, and lead) and sediment (i.e., Olsen P, Po, BD-P, residual P, copper, and nickel) in the CI model. However, some of the pollutants responded positively to dredging in water (i.e., nitrate [NO_3^--N], arsenic, cadmium, mercury, nickel, and zinc) and sediment (i.e., organic matter, TN, and NO_3^--N) in the BA model, whereas in the CI model only a few pollutants in water (i.e., NO_3^--N, chromium, and zinc)

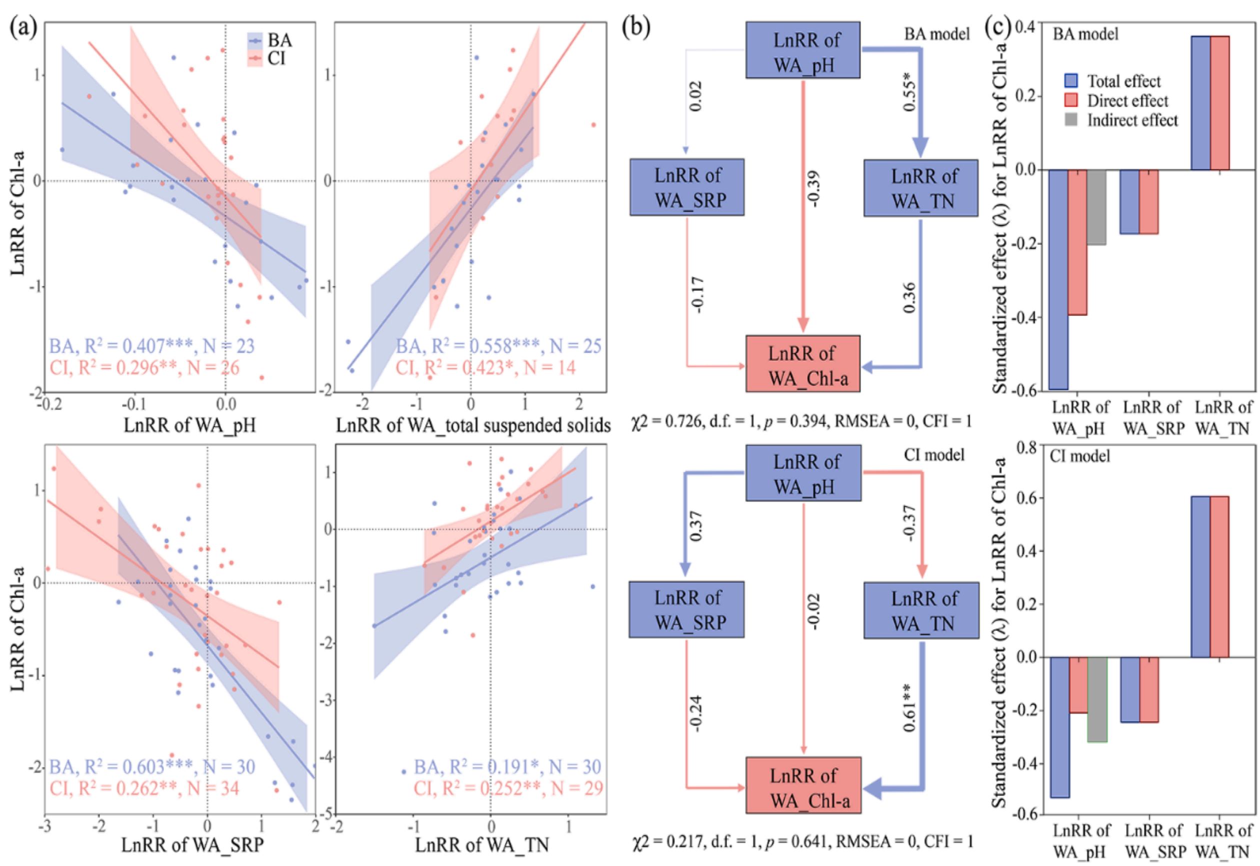

Fig. 3. Effects of water physicochemical factors on Chl-a in the BA and CI models. (a) Meta-regression reveals linkages between LnRRs of water Chl-a and water pH, water total suspended solids, water SRP, and water TN. (b) Structural equation model reflect potential relationships among LnRRs of water Chl-a, water pH, SRP, and water TN. The width of the arrows represents the strength of the standardized path coefficient. The blue and red lines represent positive and negative path coefficients, respectively. Values above the lines indicate path coefficients between two parameters. (c) Direct and indirect effects of LnRRs of water physicochemical factors on LnRR of water Chl-a. Asterisks close to numerical values denote significance (*, $p < 0.05$; **, $p < 0.01$; ***, $p < 0.001$). Abbreviations, WA, water; Chl-a, chlorophyll-a; SRP, soluble reactive phosphorus; and TN, total nitrogen.

and sediment (i.e., NO_3^--N and iron) showed a similar positive response to dredging. Consequently, 77.50% pollutants in the BA model and 84.21% pollutants in the CI model displayed negative responses to dredging, and 76.32% pollutants (e.g., water TP and NH_4^+-N) in the BA and CI models showed the same responses to dredging. These results demonstrated that dredging generally decreased pollutants in the water-sediment system, and pollutants responded uniformly to dredging in the BA and CI models.

Based on meta-regression analyses (Fig. 3a), there were notable correlations between LnRRs of Chl-a and water pH (BA model, R^2 = 0.407, $p < 0.001$, observation number = 23; CI model, R^2 = 0.296, $p < 0.01$, observation number = 26), water total suspended solids (BA model, R^2 = 0.558, $p < 0.001$, observation number = 25; CI model, R^2 = 0.423, $p < 0.05$, observation number = 14), water SRP (BA model, R^2 = 0.603, $p < 0.001$, observation number = 30; CI model, R^2 = 0.262, $p < 0.01$, observation number = 34), and water TN (BA model, R^2 = 0.191, $p < 0.05$, observation number = 30; CI model, R^2 = 0.252, $p < 0.01$, observation number = 29). Based on our structural equation model (Fig. 3b), potentially close relationships were emerged among LnRRs of Chl-a, water pH, water SRP, and water TN in both BA and CI models. LnRR of water pH slightly positively affected LnRR of water SRP in both BA and CI models ($p > 0.05$), which in turn weakly negatively influenced LnRR of Chl-a ($p > 0.05$). LnRR of water pH exhibited a significantly positive effect on LnRR of water TN in the BA model ($p < 0.05$), which in turn slightly positively affected LnRR of Chl-a ($p > 0.05$). In contrast, in the CI model, LnRR of water pH showed a slightly negative effect on LnRR of water TN ($p > 0.05$), which in turn notably positively influenced LnRR of Chl-a ($p < 0.01$). The models fitted the data well, as denoted by the non-notable $\chi 2$ tests for the BA ($\chi 2$ = 0.726, d.f. = 1, p = 0.394, RMSEA = 0, and CFI = 1) and CI ($\chi 2$ = 0.217, d.f. = 1, p = 0.641, RMSEA = 0, and CFI = 1) models (Fig. 3b). LnRR of water pH showed negative effects on LnRR of Chl-a in direct and indirect manners in both BA and CI models, whereas LnRRs of water SRP and water TN separately displayed negative and positive effects on LnRR of Chl-a in direct ways (Fig. 3c). LnRR of water pH showed a larger total effect on LnRR of Chl-a compared to other factors in the BA model, whereas in the CI model, LnRR of water TN exhibited a larger total influence on LnRR of Chl-a than other factors. These results signified that both nutritional and non-nutritional factors control the extent of algal blooms in both BA and CI models.

3.2. Responses of nutrient release rate and enzymatic activity to dredging

By calculating the percentage of effect size (Fig. 2), release rates and transformation potentials of sediment nutrients in the BA (i.e., SRP, NH_4^+-N, NO_3^--N, and nitrification potential) and CI (i.e., SRP, TN, NH_4^+-N,

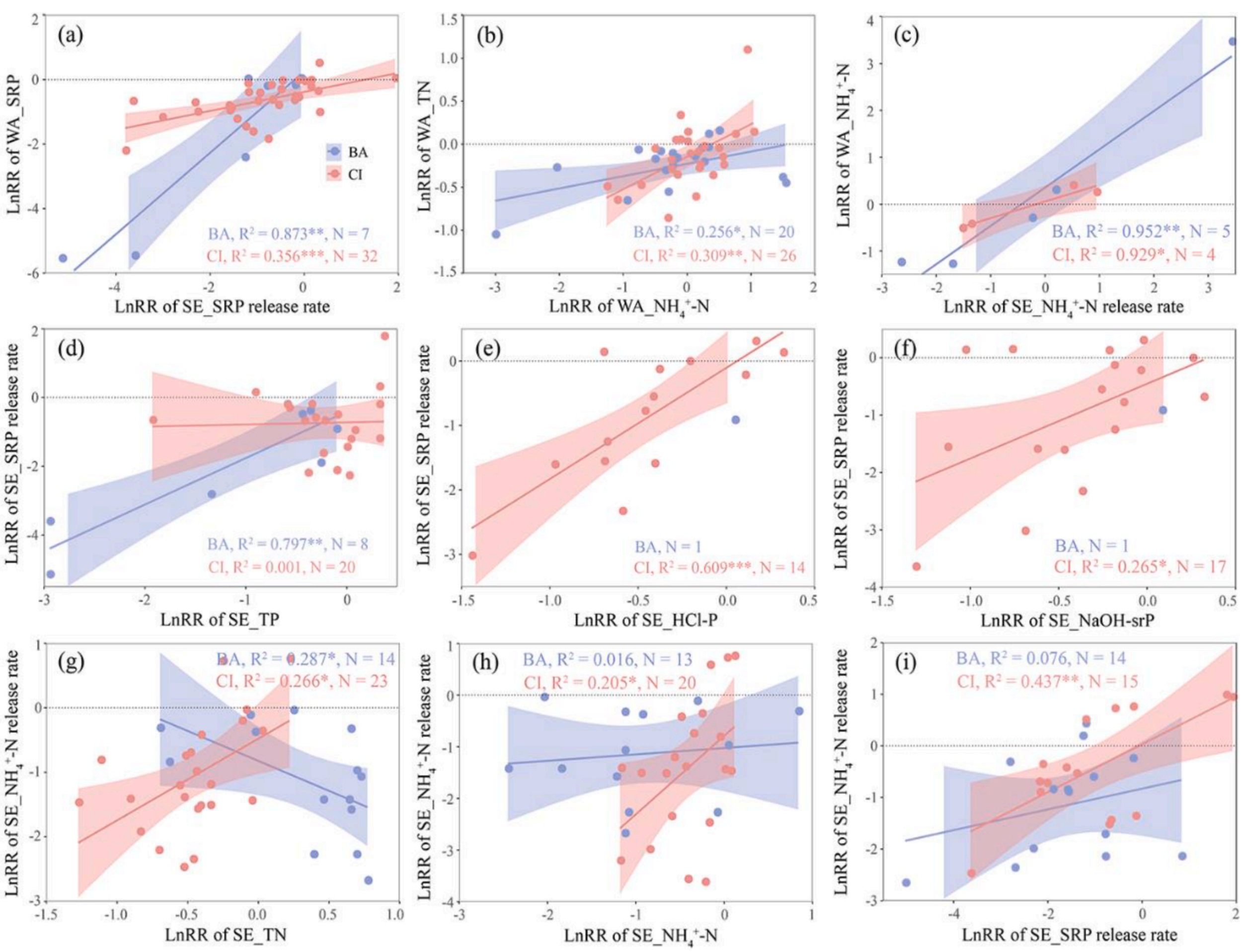

Fig. 4. Meta-regression analyses reveal linkages among water physicochemical properties, sediment physicochemical properties, and sediment nutrient release rate in the BA and CI models. Analyses identified linkages between LnRRs of water SRP and sediment SRP release rate (a), between LnRRs of water NH_4^+-N and water TN (b), and between LnRRs of water NH_4^+-N and sediment NH_4^+-N release rate (c); linkages between LnRRs of sediment SRP release rate and sediment TP (d), sediment HCl-P (e), and sediment NaOH-srP (f); and linkages between LnRRs of sediment NH_4^+-N release rate and sediment TN (g), sediment NH_4^+-N (h), and sediment SRP release rate (i). Asterisks close to numerical values denote significance (*, $p < 0.05$; **, $p < 0.01$; ***, $p < 0.001$). Abbreviations: WA, water; SE, sediment; SRP, soluble reactive phosphorus; NH_4^+-N, ammonia; TN, total nitrogen; TP, total phosphorus; HCl-P, HCl extractable phosphorus; and NaOH-srP, NaOH extractable reactive phosphorus.

nitrification potential, and denitrification potential) models displayed notably negative responses to dredging ($p < 0.05$; $p < 0.01$ or $p < 0.001$). Other parameters of sediment nutrients in the BA (i.e., TN release rate and denitrification potential) as well as CI (i.e., NO_3^--N release rate and NO_2^--N release rate) models displayed non-significant increases in response to dredging ($p > 0.05$). Sediment enzymatic activities in the BA (i.e., β−1,4-glucosidase, β−1,4-N-acetyl-glucosaminidase, and phosphatase) and CI (i.e., fluorescein diacetate) showed notably negative responses to dredging ($p < 0.01$ or $p < 0.001$; Fig. S3). Moreover, sediment enzymatic activities (i.e., leucine aminopeptidase and fluorescein diacetate) decreased insignificantly in the BA model ($p > 0.05$). These results revealed that dredging generally blocked sediment nutrient release and enzymatic activities, thus, controlling phosphorus was relatively easy compared to controlling nitrogen by using dredging.

According to our meta-regressions analyses, there were close linkages between LnRRs of sediment SRP release rate and water SRP (BA model, $R^2 = 0.873$, $p < 0.01$, observation number = 7; CI model, $R^2 = 0.356$, $p < 0.001$, observation number = 32; Fig. 4a), between LnRRs of water NH_4^+-N and water TN (BA model, $R^2 = 0.256$, $p < 0.05$, observation number = 20; CI model, $R^2 = 0.309$, $p < 0.01$, observation number = 26; Fig. 4b), and between LnRRs of sediment NH_4^+-N release rate and water NH_4^+-N (BA model, $R^2 = 0.952$, $p < 0.01$, observation number = 5; CI model, $R^2 = 0.929$, $p < 0.05$, observation number = 4; Fig. 4c). LnRR of sediment SRP release rate was notably correlated with LnRRs of sediment TP (BA model, $R^2 = 0.797$, $p < 0.01$, observation number = 8; Fig. 4d), sediment HCl-P (CI model, $R^2 = 0.609$, $p < 0.001$, observation number = 14; Fig. 4e), and sediment NaOH-srP (CI model, $R^2 = 0.265$, $p < 0.05$, observation number = 17; Fig. 4f). Additionally, LnRR of sediment NH_4^+-N release rate was significantly correlated with LnRRs of sediment TN (BA model, $R^2 = 0.287$, $p < 0.05$, observation number = 14; CI model, $R^2 = 0.266$, $p < 0.05$, observation number = 23; Fig. 4g), sediment NH_4^+-N (CI model, $R^2 = 0.205$, $p < 0.05$, observation number = 20; Fig. 4h), and sediment SRP release rate (CI model, $R^2 = 0.437$, $p < 0.01$, observation number = 15; Fig. 4i). These results suggest that sediment nitrogen and phosphorus components affect the nutrient release rate in the BA and/or CI models, and sediment SRP and NH_4^+-N release separately contribute positively to water SRP and NH_4^+-N in both the BA and CI models.

3.3. Responses of hydrobiontes to dredging

Based on the percentage of effect size (Fig. 5), in the BA model, zooplankton biomass ($p < 0.01$), phytoplankton diversity ($p < 0.05$), benthic bacterial richness ($p < 0.01$), macrophyte richness ($p < 0.05$), macrophyte diversity ($p < 0.001$), and fish density ($p < 0.001$) displayed notably positive responses to dredging. However, in the BA model (Fig. 5), planktonic bacterial richness ($p < 0.01$), planktonic bacterial diversity ($p < 0.01$), zoobenthos richness ($p < 0.01$), zoobenthos

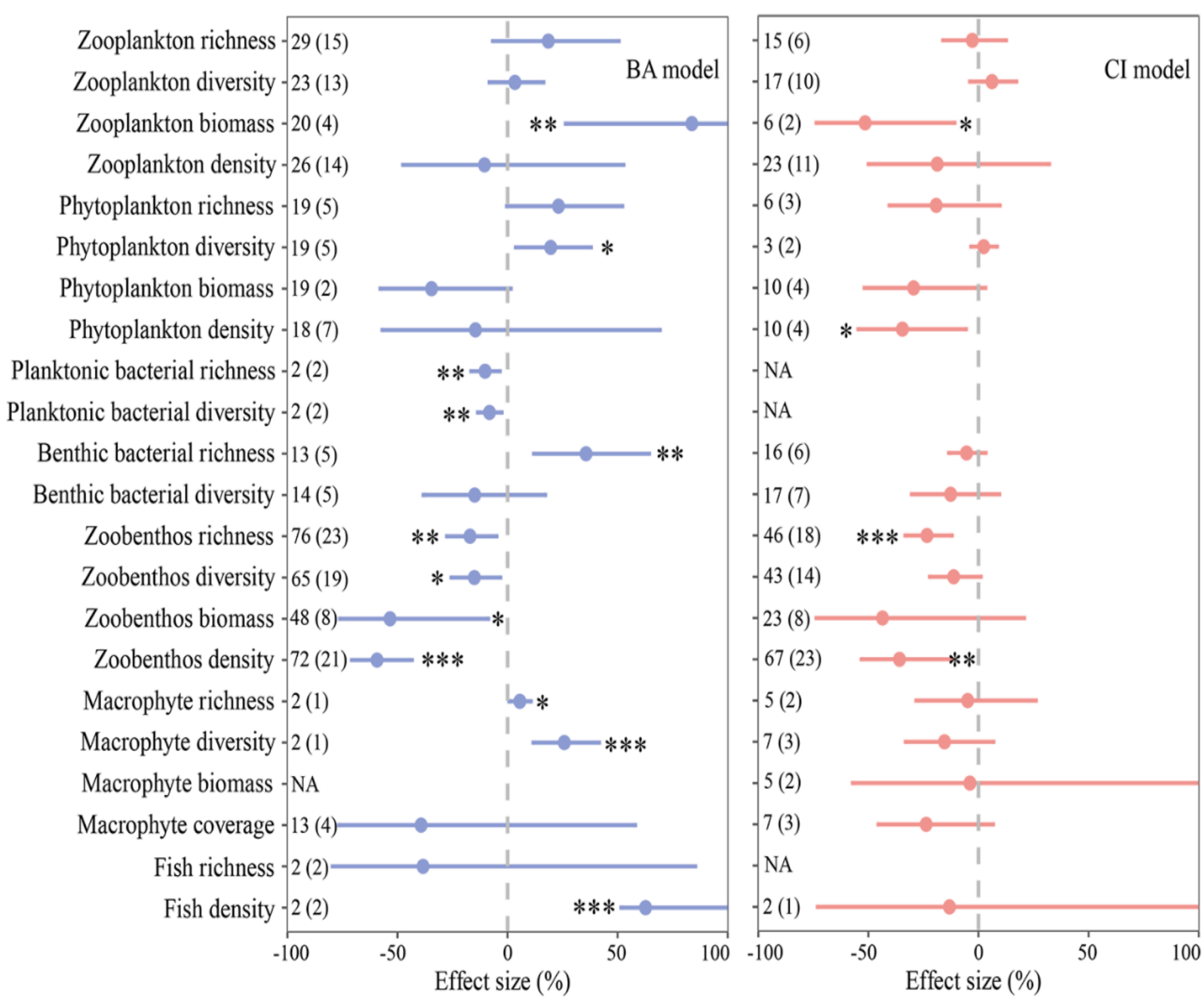

Fig. 5. Effects of dredging on attributes (i.e., richness, diversity [Shannon index], biomass, and density) of hydrobiontes in the BA (before vs. after) and CI (control vs. impact) models. Values represent effect sizes ± 95% confidence intervals, and asterisks denote significance (*, $p < 0.05$; **, $p < 0.01$; ***, $p < 0.001$). Dashed lines represent lnRR = 0 and numbers within and outside the parentheses separately denote the number of articles and studies. Different values (i.e., mean values represented by blue and red dots) of "effect size (%)" for single parameter denote that there is difference between BA and CI models.

diversity ($p < 0.05$), zoobenthos biomass ($p < 0.05$), and zoobenthos density ($p < 0.001$) showed significantly negative responses to dredging. In contrast, in the CI model (Fig. 5), zooplankton biomass ($p < 0.05$), phytoplankton density ($p < 0.05$), zoobenthos richness ($p < 0.001$), and zoobenthos density ($p < 0.01$) exhibited remarkably negative responses to dredging. Consequently, 57.14% of hydrobiont attributes in the BA model and 89.47% in the CI model responded negatively to dredging, and 61.11% of hydrobiont attributes (e.g., zoobenthos richness and zoobenthos density) showed the same responses to dredging in both BA and CI models. These results indicate that dredging causes both positive and negative effects on hydrobiontes in the BA model, but more negative effects in the CI model.

Meta-regressions revealed notable correlations between LnRRs of zooplankton density and zooplankton richness (BA model, $R^2 = 0.440$, $p < 0.001$, observation number = 24; Fig. 6a), zooplankton diversity (BA model, $R^2 = 0.232$, $p < 0.05$, observation number = 18; Fig. 6b), and zooplankton biomass (BA model, $R^2 = 0.908$, $p < 0.05$, observation number = 4; Fig. 6c). Additionally, notable correlations existed between LnRRs of phytoplankton density and phytoplankton richness (BA model, $R^2 = 0.506$, $p < 0.01$, observation number = 15; CI model, $R^2 = 0.796$, $p < 0.05$, observation number = 5; Fig. 6d), phytoplankton diversity (BA model, $R^2 = 0.368$, $p < 0.05$, observation number = 15; Fig. 6e), and phytoplankton biomass (BA model, $R^2 = 0.906$, $p < 0.001$, observation number = 7; Fig. 6f). Similarly, significant correlations were found between LnRRs of zoobenthos density and zoobenthos richness (BA model, $R^2 = 0.396$, $p < 0.001$, observation number = 64; CI model, $R^2 = 0.572$, $p < 0.001$, observation number = 40; Fig. 6g), zoobenthos density (BA model, $R^2 = 0.106$, $p < 0.05$, observation number = 47; Fig. 6h), and zoobenthos density (BA model, $R^2 = 0.301$, $p < 0.01$, observation number = 30; CI model, $R^2 = 0.177$, $p < 0.05$, observation number = 21; Fig. 6i). LnRR of zooplankton density was significantly correlated with LnRR of phytoplankton density in the BA ($R^2 = 0.486$, $p < 0.05$, observation number = 10) and CI ($R^2 = 0.879$, $p < 0.05$, observation number = 4) models (Fig. 6j). These results suggest that there are potentially close linkages between density and α-diversity and biomass of hydrobiontes in dredging-disturbed environments.

4. Discussion

4.1. Dredging shows positive effects on decreasing pollutants and controlling algal blooms

Dredging displayed positive influences on decreasing pollution by nutritional, heavy metal, and organic pollutants indicated by decreases in 77.50% and 84.21% of the pollutants in the BA and CI models, respectively (Fig. 2, Fig. S2). Our meta-analysis results comprehensively deciphered that dredging seems to be a preferred ecological restoration approach to control pollutants in water-sediment systems despite some unwanted, negative effects of dredging in some cases (Gao et al., 2019; Kang et al., 2023). Whether dredging shows positive or negative effects mainly depends on dredging intensity (Hopkins et al., 2011), content of pollutants in different sediment layers (Sun et al., 2019), and seasonal climate (Chen et al., 2016). For instance, high intensity of dredging can remove large amounts of pollutant-containing sediments, and dredging performance of pollutant removal is basically better in autumn-winter than in summer, i.e. the non-growing season (Chen et al., 2016). However, when pollutants are less abundant in the upper than in the deeper sediment layers (e.g., 0–20, 40–60, and 80–100 cm; Sun et al., 2019; Yin et al., 2021), decent amounts of pollutants can be transferred from the lower sediment layer to the water column although the upper sediment layer is disturbed by dredging (Yu et al., 2019). Therefore, dredging is often adopted jointly with barrier reagents (e.g., lanthanum-modified bentonite) to reduce pollutant availability (Lürling et al., 2017; Li et al., 2023). Additionally, warming (Zhong et al., 2022), acidification

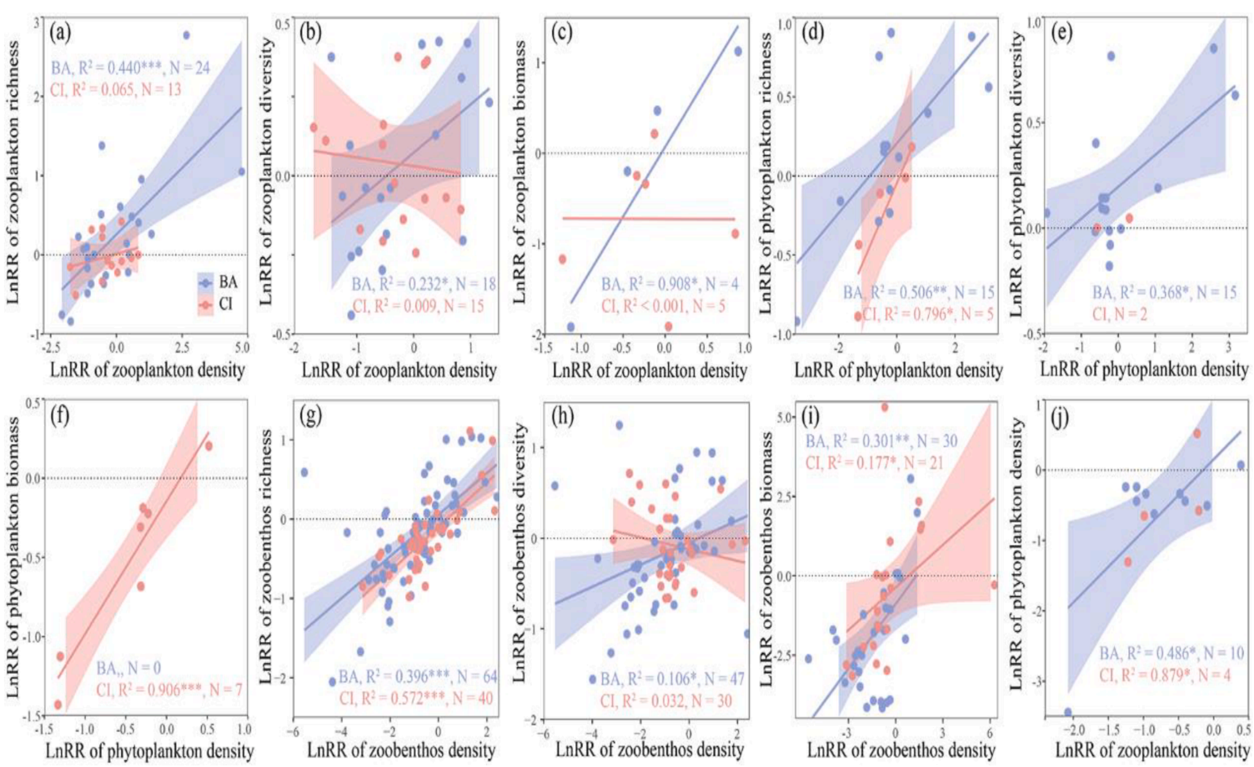

Fig. 6. Meta-regression analyses reveal linkages between density and richness, diversity, and biomass of hydrobiontes in the BA and CI models. Analyses identified linkages between LnRRs of zooplankton density and zooplankton richness (a), zooplankton diversity (b), and zooplankton biomass (c); linkages between LnRRs of phytoplankton density and phytoplankton richness (d), phytoplankton diversity (e), and phytoplankton biomass (f); linkages between LnRRs of zoobenthos density and zoobenthos richness (g), zoobenthos diversity (h), and zoobenthos biomass (i); and linkage between LnRRs of zooplankton density and phytoplankton density (j). Asterisks close to numerical values represent significance (*, $p < 0.05$; **, $p < 0.01$; ***, $p < 0.001$).

(Wan et al., 2022), and sudden disturbance (e.g., wind blow; Chen et al., 2021) weaken restoration performance of dredging. To achieve the optimal, positive effect on pollutant removal in practical restoration operations, it is necessary to (i) estimate levels of multiple pollutants in different sediment layers first and (ii) subsequently implement dredging with appropriate intensity and depths under appropriate conditions (e.g., right season and reduced disturbance impacts).

Pollutants can converge into aquatic ecosystems via water circulation and sedimentation both in are in close dependency on physicochemical processes and potentially enriches sediment nutrient levels (Bruno et al., 2020). Therefore, dredging can directly remove nitrogen- and phosphorus-containing sediment layers by using smart machines (Zhong et al., 2022). Non-nutritional factors (e.g., pH) affect maintenance and turnover of nutrients via biogeochemical process (Deemer et al., 2023), and nutritional (i.e., water SRP and TN) and non-nutritional (i.e., water pH and total suspended solids) pollutants were closely linked to algal blooms in the BA and CI models (Fig. 3). This seems reasonable as superfluous nitrogen and phosphorus generally nourish the growth and development of algae (Severiano et al., 2023; Feng et al., 2024), whereby changes in microenvironments (e.g., pH and light) affect algal growth (Wágner et al., 2018). Additionally, dredging can block nutrient release from the sediment to the water column as indicated by decreases in release rates of sediment nitrogen and phosphorus (Fig. 2; Yu et al., 2016; Chen et al., 2021). This result might be primarily due to decreases in sediment enzymatic activities and pronounced changes in microenvironments (Wan et al., 2020). Microorganisms are responsible for nutrient turnover (e.g., nitrification, denitrification, and phosphorus mineralization and solubilization) via enzymatic reactions (Hill et al., 2006), and microenvironmental changes (e.g., pH, oxygen level, and nutrient availability) affect transformation and release of sediment nutrients mediated by microorganisms (Oldenborg and Steinman, 2019; Ren et al., 2021). It is worth mentioning that content and release rate of all phosphorus fractions rather than all nitrogen components responded negatively to dredging in both the BA and CI models (Fig. 2), suggesting that controlling nitrogen by dredging in aquatic ecosystems is more difficult than phosphorus. This result is reasonable because atmospheric nitrogen can enter water via microbial nitrogen fixation, and nitrogen components (e.g., nitrate, nitrite, and ammonia) are subjected to multiple metabolic pathways and often transform mutually (Ehrenfels et al., 2023). Therefore, dredging mitigates algal bloom might via two aspects: (i) directly reducing nitrogen and phosphorus and thus change algal growth condition and (ii) indirectly weakening sediment nitrogen and phosphorus release by changing microenvironments.

4.2. Different hydrobiont responses to dredging

Richness (i.e., species number), diversity (i.e., Shannon index), biomass, and density of hydrobiontes (i.e., zooplankton, phytoplankton, zoobenthos, bacteria, macrophyte, and fishes) responded differently to dredging in both the BA and CI models (Fig. 5), whereby dredging showed negative effects on 57.14% and 89.47% of hydrobiont attributes in the BA and CI models, respectively. These results are reasonable and might be related to multiple aspects. On the one hand, dredging results in ecological drift of aquatic organisms by directly removing sediments and plants (Wan et al., 2021a), which in turn decreases richness, diversity, biomass, and density of some hydrobiontes. For instance, dredging lead to a loss of macroinvertebrates (Grygoruk et al., 2015), and lakeshore vegetation removal by dredging decreases diversities of water and sediment bacteria (Wan et al., 2024). On the other hand, losses of specific hydrobiontes at low trophic levels affect the stability of aquatic food web (Weitere et al., 2018) after dredging, which in turn decreases richness, diversity, biomass, and/or density of predators of other hydrobiontes. In contrast, hydrobiontes at low trophic levels can maintain and even increase their richness, diversity, biomass, and/or density when there are losses of their predators (Antiqueira et al., 2022). Some aquatic organisms with a relatively strong movement potential can escape from disturbed environments (e.g., imposed by dredging event or climatic change) and therefore decrease local hydrobiont pools (Braun et al., 2023), whereas hydrobiontes (e.g., zooplankton and fishes) can also remigrate to a more livable habitat (e.g., less pollution and abundant oxygen) after dredging and therefore enrich the local hydrobiont pools (Ko et al., 2020). Additionally, biological invasion from terrestrial ecosystem to aquatic ecosystem in dredged regions might also affect local hydrobiont community structure via changing species interaction relationships (Breviglieri and Romero, 2017).

Close linkages between density and richness, diversity, and biomass of hydrobiontes (i.e., zooplankton, phytoplankton, and zoobenthos) were found in the BA and/or CI models (Fig. 6), and earlier studies report synchronous changes in hydrobiont density, richness, diversity, and biomass in response to dredging (Ko et al., 2020; Pledger et al., 2021). These findings might support the density-dependent theory, namely, a sufficiently high species density is the foundation for maintaining richness, diversity, and biomass even at disturbed conditions (Hannan and Carroll, 1995; Azizan et al., 2023). Sufficiently high species density of a population/community can ensure that individuals obtain enough resources and living space in the fight of competitors of other species, and sudden death of even high fractions of individuals of the respective population/community by disturbance cannot cause extinction of the entire population/community (Wilcox and Elderd, 2003). For example, although dredging decreases bacterioplankton abundance (Wan et al., 2020), the decrease in bacterioplankton abundance was accompanied by only slight decreases in taxon diversities of the bacterioplankton (i.e., rare vs. abundant and generalist vs. specialist taxa) (Wan et al., 2021b; Yang et al., 2023). Positive linkages were found between zooplankton density and phytoplankton density in the BA and CI models (Fig. 6j), suggesting a synchronized changes between zooplankton and phytoplankton in the dredging-disturbed system. Additionally, phytoplankton is one of prioritized choices of zooplankton in the food web (Lampman and Makarewicz, 1999), and external and/or internal inputs of zooplankton to water can mitigate algal blooms (Nam et al., 2016).

4.3. Publication bias test and ecological conservation intimation

By investigating changes in LnRRs in the funnel plots using the trim-and-fill method (Hou et al., 2020), 91.899% and 93.42% factors (e.g., pollutants and hydrobiontes) in the BA and CI model, respectively, showed insignificant changes (Table S8), suggesting that the influences of publication biases are insignificant. By estimating linkages between LnRRs and publication time, 81.08% and 88.73% correlations in the BA and CI model, respectively, were insignificant (Table S9). Taken together, our supplemental analyses revealed that our results were robust.

We constructed a conceptual model to summarize our findings (Fig. 7). Generally, dredging shows negative effects on algal blooms, pollutants, nutrient fluxes, and hydrobiontes in the BA and CI models (Fig. 7a), and most tested factors belonging to pollutants and hydrobiontes respond uniformly to dredging in the BA and CI models (Fig. 7b). Density, richness, diversity, and biomass of specific hydrobiontes (i.e., zooplankton, phytoplankton, and zoobenthos) potentially obey the density-dependent theory in dredging-disturbed conditions (Fig. 7c), which might be seen as a breakthrough point to adjust hydrobiontes to further mitigate algal blooms. Pollutants and hydrobiontes showed different response ratios (i.e., LnRR) to dredging between BA and CI models. This is reasonable because BA (i.e., before vs. after) and CI (control vs. impact) are two distinctly different models, and pollutant loading, hydrobionte's migration, and interaction between hydrobiontes and environments are different in disturbed sites and non-disturbed sites (Molozzi et al., 2013; Johannesen et al., 2017). For instance, dredged sites rather than non-dredged will be further protected to reduce external pollutant loading, and there might be a small

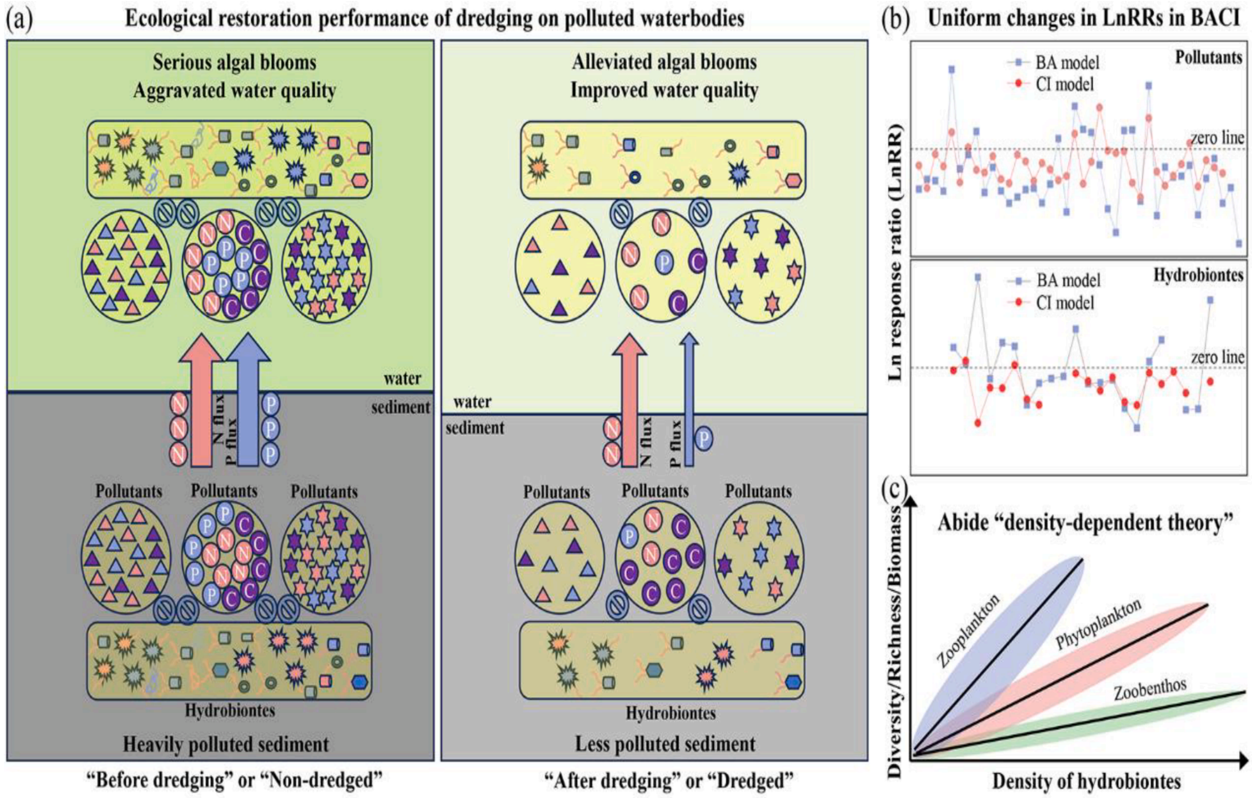

Fig. 7. Conceptual mode showing ecosystem consequence of dredging operation for restoration purposes. (a) Ecological restoration performance of dredging on polluted waterbodies, e.g., decreasing pollutants and hydrobiontes, improving water quality, mitigating algal blooms, and weakening sediment nitrogen and phosphorus fluxes. Pollutants include nutritional, heavy metal, and organic pollutants. Hydrobiontes include zooplankton, phytoplankton, planktonic bacteria, benthic bacteria, zoobenthos, macrophyte, and fishes. Symbols and colors denote different pollutants or hydrobiontes. (b) Uniform changes in LnRRs of pollutants and hydrobiontes in the BA and CI models. (c) Close linkages between density and diversity represented by Shannon index, richness, and biomass of hydrobiontes, which potentially obey the "density-dependent theory".

difference in the BA model and a large difference in the CI model in terms of a specific environmental parameter. By estimating linkages between LnRRs of tested factors (e.g., pollutants and hydrobiontes) and recovery time after dredging, 87.84% and 92.96% of the correlations in the BA and CI model, respectively, were insignificant (Table S10), which supports that dredging can achieve long-term effectiveness on ecological restoration (Sun et al., 2019). People are satisfied with dredging restoration performance for mitigating algal blooms and reducing pollutants, however, at the same time, people are also worried about hydrobiont diversity losses following dredging. This should not be an anxiety because dredging can enhance habitat diversity despite temporary biodiversity loss (Stryjecki et al., 2021) and recovery of habitats can increase species diversity and promote community complexity in the long-term run (Gawecka and Bascompte, 2023; Hernández-Carrasco et al., 2023). When ecological restoration performance of some specific aquatic ecosystems (e.g., artificial wetlands, urban lakes, and small rivers) results in improved water quality with less pollutants and harmful hydrobiontes, dredging seems to constitute a powerful mitigation measure. In particular, when combined with other restoration approaches (e.g., usage of blockers and setting of vegetation-based ecological floating islands), dredging holds a great ecological restoration potential assuming local governments have sufficient funding and high demands for long-lasting positive results.

5. Conclusions

Our findings have important implications for comprehending ecological restoration performance of dredging in various aquatic ecosystems at the global scale. Our findings, based on a global meta-analysis, show generally positive effects of dredging via decreasing pollutants and mitigating algal blooms as well as blocking sediment nutrient fluxes. Attributes (i.e., richness, diversity, biomass, and density) of hydrobiontes generally respond negatively to dredging despite positive responses of some hydrobiontes. These findings might be verified or challenged in future case studies. Yet, our findings are suited to guide the formulation of environmental protection policies and the implementation of dredging as a cost-efficient mitigation measure to improve water quality and restore aquatic ecosystems. More work must be done to unveil the underlying ecological mechanisms of dredging restoration by using improved biochemical, molecular, and microscopic techniques in the future.

Data availability

Source data for this meta-analysis and relevant reference list for meta-analysis are available from the Dryad Digital Repository https://doi.org/10.5061/dryad.w6m905r02. Data will be made available on request.

CRediT authorship contribution statement

Wenjie Wan: Writing – original draft, Software, Methodology, Funding acquisition, Data curation, Conceptualization. **Hans-Peter Grossart:** Writing – review & editing, Funding acquisition, Conceptualization. **Qinglong L. Wu:** Writing – review & editing. **Xiang Xiong:** Methodology, Formal analysis. **Wenke Yuan:** Formal analysis. **Weihong Zhang:** Software. **Quanfa Zhang:** Resources, Formal analysis. **Wenzhi Liu:** Writing – review & editing. **Yuyi Yang:** Writing – review &

editing, Funding acquisition, Conceptualization.

Declaration of competing interest

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

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.watres.2025.123506.

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