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Effects of Dredging Season on Sediment Properties and Nutrient Fluxes across the Sediment–Water Interface in Meiliang Bay of Lake Taihu, China

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Abstract: The influence of dredging season on sediment properties and nutrient fluxes across the sediment–water interface remains unknown. This study collected sediment cores from two sites with different pollution levels in Meiliang Bay, Taihu Lake (China). The samples were used in simulation experiments designed to elucidated the effects of dredging on internal loading in different seasons. The results showed that dredging the upper 30 cm of sediment could effectively reduce the contents of organic matter, total nitrogen, and total phosphorus in the sediments. Total biological activity in the dredged sediment was weaker ($p < 0.05$) than in the undredged sediment in all seasons for both the Inner Bay and Outer Bay, but the effect of 30-cm dredging on sediment oxygen demand was negligible. Dredging had a significant controlling effect on phosphorus release in both the Inner Bay and Outer Bay, and soluble reactive phosphorus (SRP) fluxes from the dredged cores were generally lower ($p < 0.05$) than from the undredged cores. In contrast, NH_4^+-N fluxes from the dredged cores were significantly higher ($p < 0.05$) than from the undredged cores in all seasons for both sites, this indicates short-term risk of NH_4^+-N release after dredging, and this risk is greatest in seasons with higher temperatures, especially for the Inner Bay. Dredging had a limited effect on NO_2^--N and NO_3^--N fluxes at both sites. These results suggest that dredging could be a useful approach for decreasing internal loading in Taihu Lake, and that the seasons with low temperature (non-growing season) are suitable for performing dredging projects.

Keywords: dredging season; inorganic nitrogen; soluble reactive phosphorus; benthic fluxes; Taihu Lake

1. Introduction

Over the past two decades, many lakes around the world have experienced eutrophication, which has caused serious environmental problems such as frequent algal bloom outbreaks, black and foul water, and pollution [1–4]. Depending on the situation, sediment can serve as a source or sink of pollutants, when sediment acts as a source, sediments are considered important sources of nutrient release to the water column, constantly replenishing nutrients and promoting the formation of algal blooms [5,6]. Studies have shown, even under the conditions of control of external sources, that eutrophication could be maintained for many years because of internal loading [7]. Many techniques are used to control internal nutrient loading, such as in situ capping [8], algae salvage [9], ecological remediation [10], and sediment dredging [11]. As a common ecoengineering technology, sediment dredging is used in many lakes to control internal nutrient loading and to rebuild the natural ecological system [12–15].

Dredging is one of the few options available for improving the ecological balance of lakes by removing contaminated sediments. However, based on available evidence, there are questions regarding the efficacy of the approach and the degree to which dredging results in reduced risk to human health and the environment [16]. Many studies have confirmed the positive effects of dredging on internal nutrient control [17–20]. However, sediment dredging is not always successful in controlling sediment nutrient release [12,21]. Thus, dredging success remains variable [14], and the mechanism underlying the observed effects on nutrient cycling remains puzzling. Previous studies have shown that the controlling effect of sediment dredging on nutrients is affected by many factors, such as the dredging technique, dredging depth, external control, and environmental conditions of the lake itself [17,21–24]. Recently, Chen et al. [25] reported that different dredging seasons have different effects on the prevention of black and odorous formations in the water column, and that dredging to prevent the formation of algae-induced black blooms could be more effective in winter than other seasons. However, the influence of dredging season on internal nutrient control has rarely been reported.

As a physical engineering measure, sediment dredging destroys the original and stable sediment–water interface, and it exposes buried sediment. This newly exposed sediment forms a new and unbalanced sediment–water interface [14,17,18]. Sediment dredging changes the physical, chemical and biological properties of sediments considerably, affecting both the cycling process of nutrients in the sediments and the nutrient flux at the sediment–water interface [19,20,26]. Additionally, nutrient cycling in a lake ecosystem has obvious seasonal characteristics [27–29], which include changes both in the nutrients in porewater and/or sediments and in nutrient fluxes across the sediment–water interface [30–33]. Currently, the selection of dredging season in China, as in many other countries, is random, and the influence of seasons on the effects and effectiveness of dredging is not considered.

In this study, we hypothesized that nutrient fluxes at the sediment–water interface would be different before and after dredging, and we hypothesized further that dredging season would have different effects on internal loading control. We collected sediment cores from two sites with different pollution levels in Meiliang Bay Taihu Lake (China). We used these samples in simulation experiments designed to elucidate the effects of dredging on internal loading in different seasons. The study had three primary objectives: (1) to understand the influence of sediment dredging on sediment properties; (2) to evaluate the effect of dredging season on nitrogen and phosphorus release and recommend the optimum season for dredging with minimal environmental risk; and (3) to study the effectiveness of dredging on internal loading at different pollution levels in sediment. The results could be used to evaluate potential dredging plans and to improve the decision-making process associated with dredging engineering.

2. Materials and Methods

2.1. Site Description and Sampling

Taihu Lake is a typical large shallow lake with a surface area of 2338 km^2 [34]. Taihu Lake is the third-largest freshwater lakes in China, and it serves as an important source of drinking water for surrounding cities, for example, Wuxi and Suzhou. Meiliang Bay is one of the most eutrophic bays in the northern part of Taihu Lake, and in the past two decades, cyanobacterial blooms have occurred there during the warmer seasons [1,35]. The water supply crisis in Wuxi in May 2007 received wide attention both in China and in other countries [3]. In attempting to reconstruct the lake ecosystem, the Chinese government has adopted many different measures since 2007, for example, external controls, ecological water diversion, salvaging blue-green algae, ecological restoration, and sediment dredging [9,36]. The latter is considered an important measure with which to control the internal loading in Taihu Lake. The area of Taihu Lake that was dredged was about 93.65 km^2, and Meiliang Bay was pinpointed as a key area in the dredging plan of Taihu Lake [9].

Two sites (Inner bay and Outer Bay) with different pollution levels in Meiliang Bay (Figure 1) were selected in this study. Inner Bay is close to the city of Wuxi, Previously, untreated sewage was discharged into Meiliang Bay through the Liangxi River, a main river running through Wuxi, Furthermore, the surrounding land is a scenic area, and thus, Inner Bay has been subject to intensive use and it has suffered because of human activities. Outer Bay is located in the inlet of Meiliang Bay, the water and sediment quality at this location are affected primarily by Taihu Lake. The nearby land use is mainly orchards and grassland, and vegetation coverage is high. Therefore, Outer Bay is affected primarily by nonpoint source pollution [37].

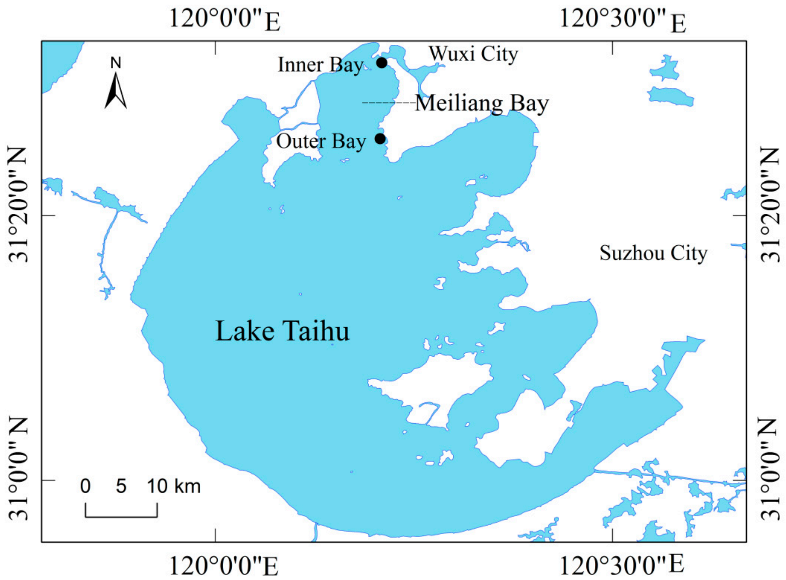

Figure 1. Sampling sites in Taihu Lake, China.

Sampling was conducted at both sites in January (winter), April (spring), July (summer), and October (autumn) through 2013. Sampling was conducted usually in the middle of each month, and 36 sediment cores was collected using a gravity corer (9 cm diameter, 70 cm length) from the two sites in each season. These sediment cores were used to test nutrient flux, sediment oxygen demand and total microbial activity at the sediment-water interface. In January (winter) 2013, one sediment core was collected at each site for analysis of basic physical and chemical properties of the sediments. All sediment cores collected in this study were >50-cm long, and both the top and the bottom of each core was plugged with a rubber stopper. In the sampling and transportation processes, care was taken to avoid disturbance and destruction of the sediment structure. Near-bottom water was sampled in situ at each site and stored in acid-cleaned polyethylene buckets. The water temperature at each site was measured using a multiparameter water quality instrument (YSI, Yellow Springs, OH, USA). All the sediment cores and water samples were then returned to the laboratory for processing.

2.2. Laboratory Microcosm Experiment

For dredging simulation and the laboratory microcosm experiment, we have adopted the procedure described by Kleeberg et al. [17] and Reddy et al. [18]. In the laboratory, overlying water was siphoned off from each core. To simulate dredging, the uppermost 30 cm of sediments in six undisturbed cores from the two sites was skimmed using a suction pump. The remaining sediment cores (9 cm diameter, 20 cm length) were then transferred to cleaned plexiglass tubes to serve as the dredged cores (Supplementary Figure S1). Six additional undisturbed cores from the two sites were adjusted to lengths of 20 cm to serve as the undredged (control) cores, specifically, the excess sediment layers at the bottom were removed, the remaining upper 20 cm sediment layers were then transferred to cleaned plexiglass tubes to serve as the undredged cores (Supplementary Figure S1). There were three replicates for both undredged and dredged treatments. Filtered water from in situ water samples was injected into both the dredged and the undredged cores using the siphon method, and the depth of the overlying water was maintained at 20 cm. Subsequently, the sediment of the cores was wrapped in silver paper to avoid light. The undredged and dredged cores were then transferred to a water bath and incubated at the in situ water temperature (±2 °C). Water samples (50 mL), taken from the incubated cores at designated intervals (0, 12, 24, 36, 48, 60, and 72 h), were filtered through 0.45-μm syringe filters and analyzed for soluble reactive phosphorous (SRP) and inorganic nitrogen. Filtered site water (50 mL) was added to each core immediately after each sampling to maintain water quantity.

The exchange rates of nutrients across the water–sediment interface were calculated according to the following equation [38]:

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

where F_i represents the exchange rate during the three days (mg $m^{-2}d^{-1}$); V represents the volume of overlying water in the sediment column (L); C_0, C_n and C_{j-1} represent the nutrient concentration (mg L^{-1}) at the beginning, on day n and on day $j-1$, respectively; C_a represents the nutrient concentration of the water used to compensate for the sampled water (mg L^{-1}), V_{j-1} represents the water volume sampled each time (L); S represents the area of the sediment column (m^2) and T represents the incubation time (d).

2.3. In Situ Porewater Sampling

Porewater equilibrators, also referred to as peepers [39], were used to obtain the porewater profiles at both sites in different seasons. Each peeper had 36 vertically disposed dialysis cells machined to 1 cm resolution (Supplementary Figure S2). The structure and operating principle of a peeper has been described in detail by Jacobs [40]. When performing interstitial water sampling, the peepers were placed at the sediment-water interface by a self-made device and fixed on a steel shelf by rope at each site, part of the peeper inserted into the sediment, leaving a part of the peeper in the overlying water above the sediment, so that a complete sediment-water interface can be obtained. The peepers were retrieved after 15 days when the interstitial water was considered to have reached equilibrium with the peepers [41]. After retrieval of the peepers, the water of the peepers was sampled immediately, injected into vials containing appropriate fixative agents, and then stored on ice. The nutrient and Fe^{2+} contents of the porewater were analyzed within 3 h following transportation to the laboratory. Unlike the successful recovery of peepers in January 2013, peeper recovery failed in the other seasons. In order to support the results of this study, we also used unpublished porewater data of different seasons obtained at the same two sites in 2007–2008.

2.4. Analyses of Sediment and Water Characteristics

Sediment cores collected from both sites in January 2013 were used for the analysis of sediment physicochemical characteristics. Sediment cores were sliced at 2 cm intervals under anaerobic conditions (N_2 atmosphere). Sediment water content was determined by weight loss after drying the sediment samples at 105 °C for >24 h. Porosity and bulk density were measured using a cutting ring [26]. Loss on ignition (LOI) was determined after ignition at 550 °C for 6 h. Total nitrogen (TN) and organic carbon (OC) content of the sediment were analyzed using CHN elemental analyzer (CE-440, EAI, North Chelmsford, MA, USA). Total phosphorus (TP) in the sediment was analyzed by following the procedure described by Ruban et al. [42].

A further two sets of six cores (9 cm diameter) were used to investigate the effects of on sediment oxygen demand and total microbial activity. Dredging simulations were conducted according to the protocol above. There were three replicates for both undredged and dredging treatments. For the analysis of total microbial activity in the surface sediment, the surface sediments (0–2 cm layers) were sliced from both the undredged and the dredged cores immediately after dredging. The total microbial activity of the sediment was measured using fluorescein diacetate [43]. Sediment oxygen demand of the undredged and dredged cores was determined following the procedure described by Fisher and Reddy [44]. The dissolved oxygen concentrations in the water were recorded at designated intervals using an oxygen microsensor (Presens, Regensburg, Germany). The sediment oxygen demand (*SOD*) rates were calculated using the formula described by Malecki et al. [30].

$$SOD = \left\{ \left[\left(DO_i - DO_f \right) / (T \times S) \right] \times L \right\} \quad (2)$$

where SOD is sediment oxygen demand (mg cm^{-2} h^{-1}), DO_i is initial dissolved oxygen (DO) concentrations (mg L^{-1}), DO_f is final DO concentration (mg L^{-1}), T is time (h), S is surface area of core (cm^2) and L is reflood water volume (L).

Nutrient concentrations in the water samples were analyzed using a flow-injection autoanalyzer (Skalar Sanplus, Breda, The Netherlands). Fe^{2+} in the porewater was determined using the phenanthroline method with a spectrophotometer (UV-2550, Shimadzu, Kyoto, Japan) [45].

2.5. Statistical Analysis

One-way analysis of variance (ANOVA) was used to analyze the differences of nutrient in the porewater and overlying water between the four seasons and two sites. Two-way analysis of variance (ANOVA) was used to analyze the differences of nutrient fluxes, total microbial activity and SOD between undredged and dredged treatments in the four seasons. An independent *t*-test was performed to compare the differences of sediment properties between two sites and different depths. Untransformed data in all cases satisfied assumptions of normality and homoscedasticity. Statistical analysis was performed using the SPSS 20.0 statistical package, and the level of significance used was $p < 0.05$ for all tests.

3. Results and Discussion

3.1. Sediment Characterization

The porosity of sediments generally decreased with depth, whereas the contents of LOI, OC, TN, and TP decreased in the first tested depth and then increased and decreased, showing a clear peak at depths ranging from 10 to 20 cm (Figure 2). The contents of LOI, OC, and TN showed no significant differences ($p > 0.05$) between the Inner Bay and the Outer Bay samples, while the contents of TP differed ($p < 0.05$) because of the different land use types.

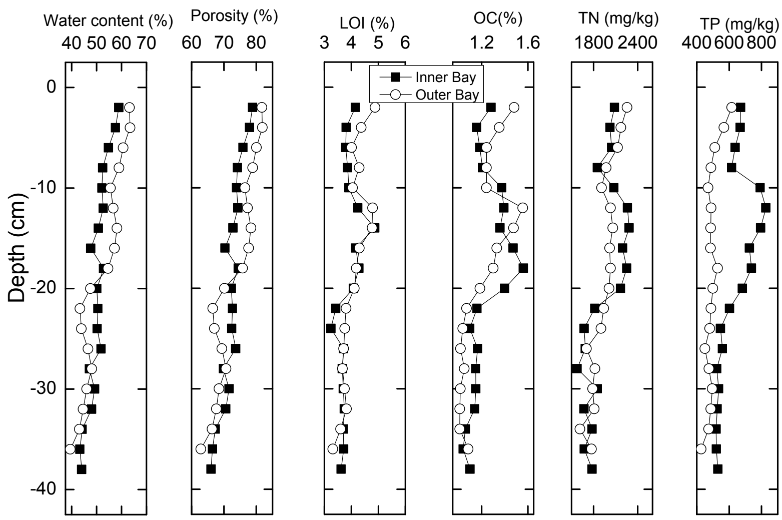

Figure 2. Vertical profiles of selected physicochemical characteristics of sediments.

In this study, we simulated a dredging depth of 30 cm. Figure 2 shows that 30 cm dredging would have considerable impact on the physicochemical properties of the sediments. The water content and porosity of the dredged sediments (buried sediment) were lower ($p < 0.05$) than the undredged sediment (surface sediment) because of the increasing compaction and dehydration of sediment with depth. Furthermore, the contents of LOI, OC, TN, and TP in the dredged sediments were significantly lower ($p < 0.05$) in comparison with the same layer in the undredged sediment. Thus, 30-cm dredging would effectively reduce the loadings of organic matter, nitrogen, and phosphorus in the sediments at both sites.

3.2. Temporal and Spatial Variations of Porewater Chemistry

Peepers are passive porewater samplers with diffusion chambers [38]. Previous studies have demonstrated that their use has good reproducibility in obtaining quality porewater profiles [40]. In this study, porewater profiles in different seasons were obtained from Inner Bay and Outer Bay in order to identify spatial and seasonal trends in biogeochemical processes. The profiles of porewater chemistry obtained by peepers are presented in Figures 3 and 4. The concentrations of NH_4^+-N, SRP and Fe^{2+} in the porewater increased with depth (Figure 3), but the concentrations of NO_3^--N and NO_2^--N decreased with depth. In general, the concentrations of NH_4^+-N and SRP in the porewater of lower sediment layer of Inner Bay were much higher ($p < 0.05$) than in Outer Bay (Figure 3). These porewater results, which are in accord with those of the sediments, show that the pollution status of Inner bay was more serious than Outer Bay.

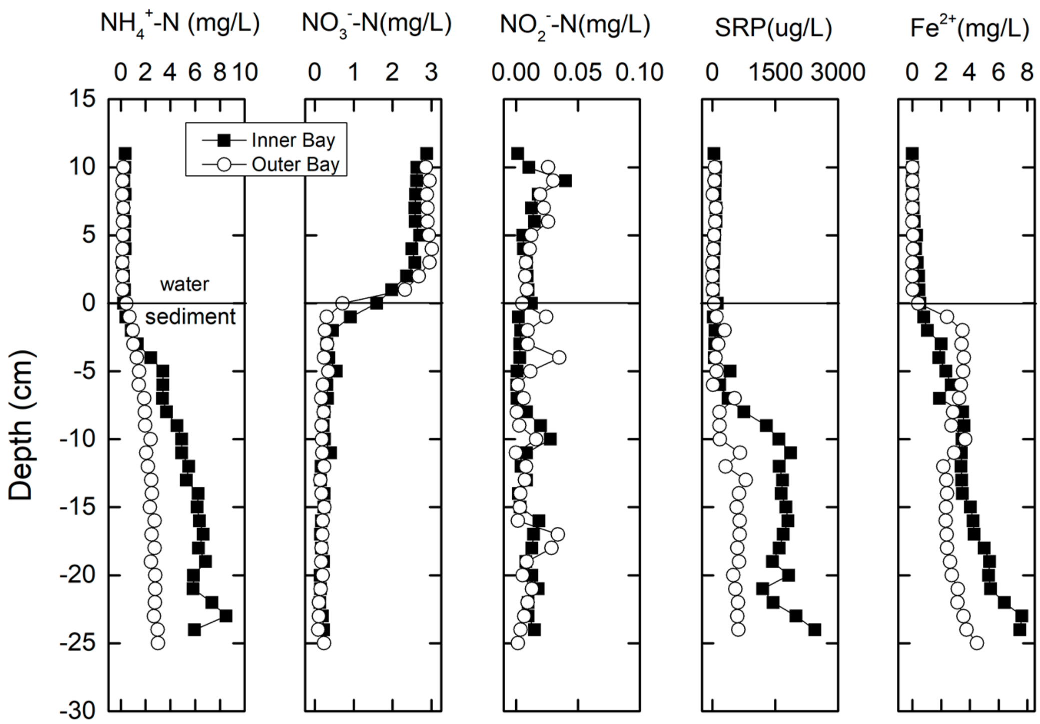

Figure 3. Depth profiles of selected chemical characteristics of porewater in Inner Bay and Outer Bay, samples were collected in January 2013.

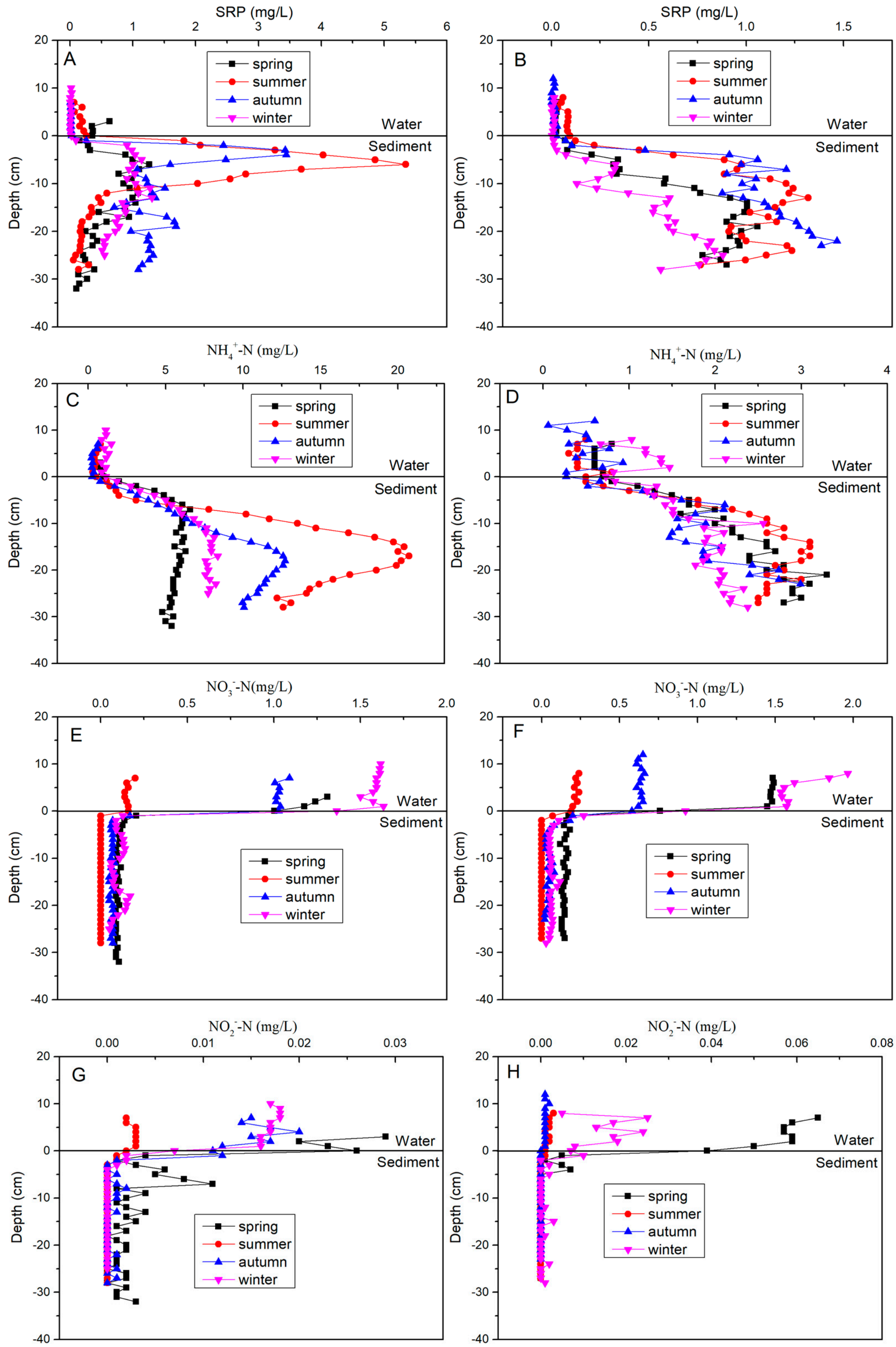

Figure 4. Depth profiles of soluble reactive phosphorus (SRP) and inorganic nitrogen in sediment porewater in different seasons from Inner Bay (**A**,**C**,**E**,**G**) and Outer Bay (**B**,**D**,**F**,**H**), samples were collected in 2007–2008.

The porewater profiles of SRP showed obvious spatial-temporal variations (Figure 4A,B). The concentrations of SRP in the upper 10 cm layer in summer and autumn were generally higher ($p < 0.05$) than in winter and spring, showing seasonal tendency at both sites. This result might be attributable to increased microbial biomass and activity in the sediments during summer and autumn, reflecting increased bioavailable organic matter present on the sediment surface [30]. The SRP profile had a peak between 0 and 15 cm in the Inner Bay, especially in summer and autumn, with high temperatures. The concentration of SRP peaked in sediment layer levels between 0 and 15 cm, which s where the maximum decomposition of organic matter, oxygen consumption, sulfide oxidation, and nutrient reduction occurs [40,46]. The concentration of SRP in porewater was generally higher ($p < 0.05$) than in the overlying water, notably by 40 and 45 times for the Outer Bay and Inner Bay respectively, indicating that sediment is an important source of phosphorus for the overlying water. In addition, the concentrations of SRP in the porewater of the Inner Bay were significantly higher ($p < 0.05$) than in the Outer Bay, especially in the surface 10 cm. These results might be attributable to the different content of TP in the sediments (Figure 2) and different sources of pollution between two sites [37].

All the peepers indicated that NH_4^+-N is the dominant form of inorganic nitrogen in porewater (Figure 4), the proportions of NH_4^+-N to inorganic nitrogen are over 93% for both sites. The profiles of NH_4^+-N showed seasonal characteristics in the Inner Bay, as concentrations of NH_4^+-N were notably higher ($p < 0.05$) in summer and autumn than in winter and spring, and there was a peak between 10- and 25-cm sediment layers. However, the profiles of NH_4^+-N in the Outer Bay had no strong ($p > 0.05$) seasonal variation (Figure 4C,D). In addition, the concentrations of NH_4^+-N in the porewater of the Inner Bay were significantly higher ($p < 0.05$) than in the Outer Bay. These results might be attributable to the different levels of pollution and different sources of pollution between the Inner Bay and Outer Bay [37]. The concentrations of NH_4^+-N in porewater were generally higher ($p < 0.05$) than in the overlying water, suggesting that sediments are an important source of nitrogen for the overlying water at both sites. Unlike NH_4^+-N, the concentrations of NO_3^--N and NO_2^--N in porewater were very low, and they were generally lower ($p < 0.05$) than the overlying water at both sites (Figure 4E,F). Furthermore, the concentrations of NO_3^--N and NO_2^--N in the overlying water showed seasonal tendencies (Figure 4G,H), i.e., the concentrations of NO_3^--N and NO_2^--N in the overlying water in winter and spring were higher ($p < 0.05$) than in summer and autumn. Concentrations of inorganic nitrogen in the water columns showed seasonal trends that are likely influenced by many factors, for example, exogenous inputs, water levels, biological absorption, and nitrogen cycling processes [47–49].

3.3. Effects of Dredging on Total Microbial Activity and Sediment Oxygen Demand

Total microbial activity and SOD are two very important characteristics of sediment. These properties have direct relationships with the decomposition process of organic matter and the transformation process of nutrients in the sediments. In this study, we evaluated the effects of dredging and of dredging season on total microbial activity and SOD. For the undredged treatment, biological activity showed significant seasonal characteristics ($p < 0.05$) in the sediments of the Outer Bay and Inner Bay (Figure 5). The values of biological activity in spring and summer were remarkably higher ($p < 0.05$) than in autumn and winter. For dredged treatment, the microbial activities of the Inner Bay sediments showed significant seasonal differences ($p < 0.05$), while those of the Outer Bay sediments showed no significant seasonal differences ($p > 0.05$). In general, biological activity in the dredged sediments was weaker ($p < 0.05$) than in undredged sediments (Figure 5). Previous research has indicated that the total number of bacteria and hydrolytic activity decrease with sediment depth [50]. In this study, the dredged sediments were buried below 30 cm, and therefore, the dredged sediments had relatively low microbial activity.

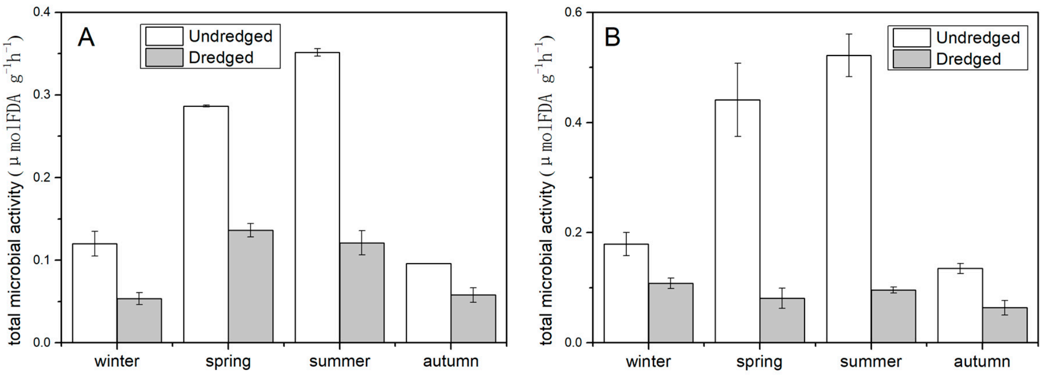

Figure 5. Total microbial activity in undredgedand dredged sediments of the Inner Bay (**A**) and Outer Bay (**B**) in January, April, July, and October 2013, Error bars represent ±1 SE of triplicate samples.

The SOD consumption rates in both the undredged and the dredged cores showed significant seasonal differences ($p < 0.05$); the highest values appeared in summer and, the lowest values appeared in winter (Figure 6). Greater rates of consumption in summer (July) are indicative of increased microbial activity due to increased water column productivity, which results in more bioavailable material than in the winter months. There was no significant difference ($p > 0.05$) in the SOD values between the undredged and dredged cores, except for Inner Bay in spring and for Outer Bay in autumn (Figure 6). This result is different from the total microbial activity in the sediment (Figure 5). Sediment oxygen consumption is attributed to chemical oxidation and microbial-mediated oxidation [30]. In the present study, we simulated a dredging depth of 30 cm, and the deep buried anaerobic sediment was exposed at the sediment–water interface after dredging. We speculate that the amount of chemical aerobic oxidation (e.g., iron oxide) in the dredged sediment was higher than in the undredged sediment. Therefore, despite the undredged sediment having higher microbial activity (Figure 5), indicating higher microbial oxidation than the dredged sediment, in most cases, no significant difference in SOD was found between the dredged and undredged cores.

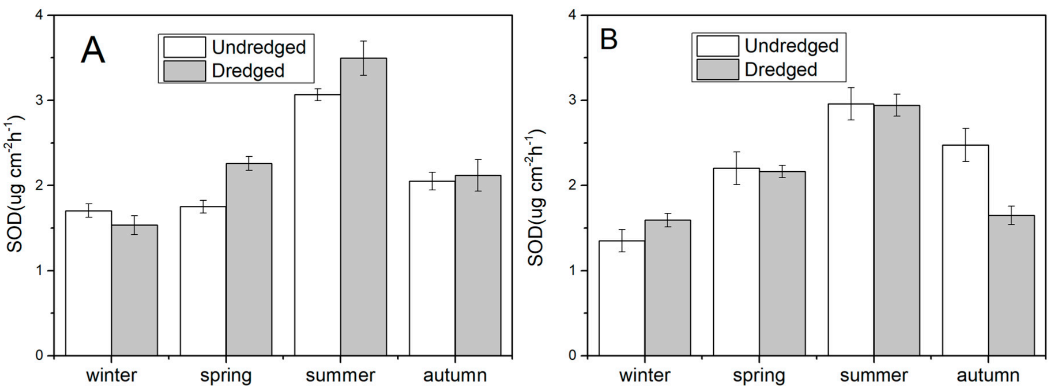

Figure 6. Sediment oxygen demand (SOD) in undredged and dredged sediments of the Inner Bay (**A**) and Outer Bay (**B**) in January, April, July, and October 2013, Error bars represent ±1 SE of triplicate samples.

3.4. Effects of Dredging on Phosphorous Flux

The fluxes of SRP had significant seasonal variation ($p < 0.05$) in the undredged cores from both the Inner Bay and Outer Bay (Figure 7A,B). Phosphorus release fluxes in summer and autumn were higher than in spring and winter; in winter, the SRP fluxes were negative, indicating that the sediment acts as a sink. The SRP fluxes in the dredged cores were negative (i.e., indicating a sink) in all seasons. It was found that the SRP fluxes in the undredged cores were significantly higher ($p < 0.05$) than in the dredged cores from both Inner and Outer Bay (Figure 7A,B). This result indicates that 30 cm dredging could effectively control phosphorus release for both sites in Meiliang Bay.

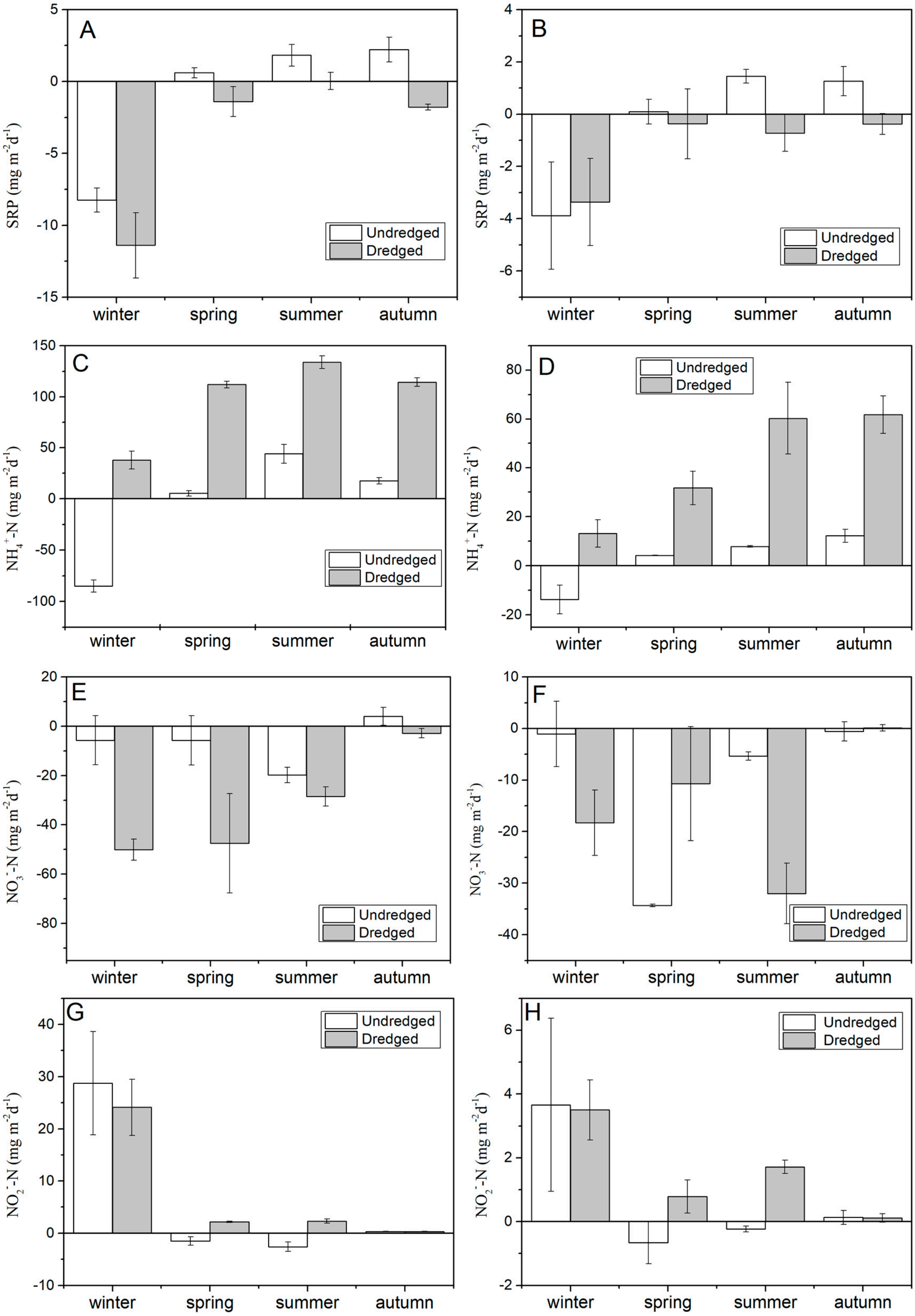

Figure 7. Fluxes of SRP and inorganic nitrogen across the sediment–water interface in undredged and dredged sediments of the Inner Bay (**A**,**C**,**E**,**G**) and Outer Bay (**B**,**D**,**F**,**H**) in January, April, July, and October 2013, Error bars represent ±1 SE of triplicate samples.

Based on simulations and fieldwork, previous studies have proven that sediment dredging can effectively control phosphorus release from sediment [20,23,51]. In the present study, we studied only the controlling effect of dredging on phosphorus release in the short term. Therefore, SRP concentration in the interstitial water, temperature, and sediment characteristics, such as porosity, were the primary factors affecting the fluxes across the sediment–water interface. The reduction in SRP release from the dredged cores in comparison with the undredged cores can be attributed to the following reasons. (1) Sediment porosity decreased considerably (Figure 2) with depth because of the increasing compaction and dehydration of the sediment. Therefore, the porosity of the dredged sediment was lower than that of the undredged cores. The lower sediment porosity of the dredged cores hindered labile phosphorus transfer from the sediments to the water column [17,52]. (2) The SRP concentration in dredged cores also decreased considerably, especially for the Inner Bay (Figure 4A). The peak of SRP in the interstitial water appeared above the 10 cm sediment layer (Figure 4A). Therefore 30 cm dredging could effectively reduce the SRP concentration in the dredged sediment, and the lower SRP concentration in the porewater of the dredged cores might present lower release potential compared with the undredged cores [20]. (3) The soluble Fe^{2+} concentration in the porewater increased considerably with depth (Figure 3). Therefore, the soluble Fe^{2+} concentration in the porewater of dredged cores would be higher than in the undredged cores. The buried and anaerobic sediments exposed at the sediment–water interface after dredging are gradually transformed to aerobic sediments. The formation of an oxic-trapped microlayer formation on the surface sediment of the dredged cores could be attributed to the oxidation of the soluble F^{2+} forms to their insoluble Fe^{3+} oxides and hydroxides [52]. Thus, the phosphorus release at the sediment–water interface of the dredged cores could be inhibited by the oxic-trapped microlayer through adsorption and coprecipitation [6,20].

The temporal effectiveness of sediment dredging in controlling internal loading is a common concern among researchers [11,53]. In this study, 30 cm dredging was shown to be effective in reducing SRP fluxes across the sediment–water interface of dredged cores (Figure 7A,B). Thus, sediment dredging might have a positive effect in reducing internal phosphorus loading just after dredging. The long-term effectiveness of sediment dredging in reducing internal phosphorus loading is affected by many factors, such as external sources, phosphorus regeneration, and the intrinsic nature of the lake itself [18,20,24]. The present study found that 30-cm dredging could reduce the contents of organic matter (LOI) and TP in the dredged sediments (Figure 2). Moreover, pools of readily mobile and labile phosphorus also became reduced [17,18,20]. Additionally, previous research has demonstrated that the lower resupply ability and the higher retention capacity of sediment after dredging could hinder labile phosphorus transfer from the sediments to the overlying water [20]. However, Liu et al. [54] reported that the accumulation of riverine suspended particulate matter in the sediment after dredging could increase internal phosphorus loading and regulate the long-term efficacy of dredging. If external inputs of nutrient loading are not controlled, external phosphorus will lead to the reversion of before dredging levels within a few years after dredging [14,51], which will affect the long-term efficacy of dredging on phosphorus control [54]. Therefore, blocking external phosphorus loading should be a precondition to achieving effective long-term results from dredging [18,51,54].

3.5. Effects of Dredging on Inorganic Nitrogen Flux

The NH_4^+-N fluxes showed significant seasonal variation ($p < 0.05$) in the undredged and dredged cores from both Inner Bay and Outer Bay (Figure 7C,D). The NH_4^+-N fluxes demonstrated lower release rates in winter and spring, and higher release rates in summer and autumn. Unlike SRP fluxes, NH_4^+-N fluxes in the dredged cores were significantly higher ($p < 0.05$) than in the undredged cores for both sites. This result indicates that NH_4^+-N release would be enhanced significantly in the short term after dredging, and this phenomenon should be noted.

In general, NO_3^- -N fluxes were mainly negative (Figure 7E,F), that is, the sediment acts as a sink for the overlying water. There were no significant differences ($p > 0.05$) in the NO_3^- -N fluxes between the dredged and undregded cores for both Inner Bay and Outer Bay (Figure 7E,F). The NO_3^--N fluxes were generally negative; therefore, the environmental risk posed by NO_3^--N release after dredging can be ignored. Unlike NO_3^--N fluxes, NO_2^--N fluxes had significant seasonal differences ($p < 0.05$) for both the undredged and dredged cores (Figure 7G,H). The NO_2^--N fluxes had higher rates in winter and lower rates in all other seasons. No significant differences ($p > 0.05$) occurred in the NO_2^--N flexes between the dredged and undredged cores for both Inner Bay and Outer Bay (Figure 7G,H). Because of the lower release rates of NO_2^--N compared with NH_4^+-N, the environmental risk posed by NO_2^--N release after dredging is lower.

The release of NH_4^+-N from the sediment–water interface would be enhanced after dredging. This phenomenon has been confirmed by previous studies [23,53,55]. This finding can be attributed to the NH_4^+-N distribution in the vertical profile. The concentration of NH_4^+-N increased with sediment depth (Figure 4C,D). Thus, after 30 cm dredging, buried sediment with higher concentrations of NH_4^+-N would be exposed at the sediment–water interface, forming a new surface layer. Compared with the undredged sediments, the lower porosity of the dredged sediments could prevent the release of NH_4^+-N. However, the higher concentration gradient at the sediment–water interface of the dredged cores would lead to significant increase in NH_4^+-N release [23,55]. Given the more serious pollution in Inner Bay, NH_4^+-N fluxes in the undredged cores from this area were bigger than from Outer Bay (Figure 7C,D). Similarly, after dredging, the NH_4^+-N fluxes in Inner Bay were also bigger than in Outer Bay (Figure 7C,D). Therefore, the environmental risk posed by NH_4^+-N release in Inner Bay is even greater than in Outer Bay.

The present study investigated the effect of dredging on the control of NH_4^+-N release for only a short period after dredging. In our previous study, we evaluated the long-term effectiveness of sediment dredging in reducing internal nitrogen release in Taihu Lake [55]. The results of that work indicated that the release rates of NH_4^+-N could be enhanced for several months after dredging, and that sediment dredging could be considered effective in controlling NH_4^+-N release four months after dredging [55]. In our opinion, the short-term effect of dredging on the release of NH_4^+-N depends on the pool sizes of NH_4^+-N in the interstitial water and labile NH_4^+-N in the sediments just after dredging. However, the long-term effect of dredging on the release of NH_4^+-N is determined by the rate of NH_4^+-N regeneration in the dredged sediment. Sediment dredging can have considerable influence on the physical, chemical, and biological properties of sediment, resulting in significant reduction of nitrogen mineralization and regeneration rates [19,53,56]. Thus, it can demonstrate effective long-term control of nitrogen release [19,23,53].

3.6. Optimum Dredging Season for Minimal Environmental Risk

Currently, in China, as in many other countries, the choice of dredging season is largely random. The implementation of a dredging engineering project depends mainly on government budgets. However, the influence of season on the effects and effectiveness of dredging is not considered in the decision-making process. This study showed that the concentrations of inorganic nitrogen and SRP in the interstitial water and the fluxes across the sediment–water interface have obvious seasonal characteristics, especially for the Inner Bay (Figures 4 and 7). The controlling effects of dredging on the benthic release of SRP and inorganic nitrogen are different in the short term after dredging. For the control of phosphorus release, under the premise of a reasonable depth of dredging, this study found 30-cm dredging in all seasons could effectively control phosphorus release. However, because of the higher SRP concentration in the interstitial water and the higher release rates of SRP in seasons with high temperatures (Figure 4A,B and Figure 7A,B), if the dredging depth were not appropriate, dredging projects conducted in seasons with high temperatures could have greater environmental risks in relation to phosphorus release after dredging. Therefore, in our view, the seasons with low temperature are appropriate for dredging projects intended to control internal phosphorus loading. However, greater attention should be given to determining the optimum dredging depth.

In the present study, we found the risk of release of NO_3^--N and NO_2^--N could be negligible because of low and/or negative fluxes across the sediment–water interface. However, the flux of

NH_4^+-N became enhanced for a short period after dredging because of the high concentration of NH_4^+-N in the porewater, even though the risk posed by NH_4^+-N release could be eliminated when a lake has been dredged for years [24,53], and despite sediment dredging being shown to be effective in controlling NH_4^+-N release from sediments in the long term [55,56], the environmental risks cannot be ignored, especially for the Inner Bay. The NH_4^+-N release rates of dredged cores were lowest in winter for both the Inner Bay and Outer Bay (Figure 7C,D), which means the risk of release of NH_4^+-N was minimized in winter. Moreover, winter is a dry season with shallow water depth, which facilitates the construction of dredging projects. In addition, the implementation of dredging projects in winter avoids the growth season of aquatic organisms, and facilitates the restoration of benthic animals and macrophytes after the implementation of dredging projects. In order to reduce internal nitrogen loading in Taihu Lake (and in other eutrophic lakes), we recommend the seasons with low temperature (non-growing season) as the seasons suitable for dredging engineering projects with minimal NH_4^+-N release risk.

It must be pointed out that we only evaluated the short-term effects of the dredging season on sediment properties and nutrient release at the sediment-water interface in this study. Longer-term studies are more conducive to understanding the environmental effects of dredging season. In our future studies, we will focus on the medium- and long-term effects of the dredging season in order to provide a reference for dredging projects. In addition, future research should also pay attention to the environmental effects of dredging depth, because the depth of dredging is not only related to the control effect of sediment dredging on the internal loading [18,57], but also directly related to the investment of dredging projects.

4. Conclusions

This study evaluated the potential effect of dredging season on sediment properties and nutrient release from the sediments in Meiliang Bay of Taihu Lake, China. The results indicated that 30 cm dredging could effectively reduce organic matter, total nitrogen and total phosphorus loading for both the Inner Bay and Outer Bay. The total biological activity in the dredged sediment was weaker than in the undredged sediment in all seasons for both Inner Bay and Outer Bay, and the effect of 30 cm dredging on sediment oxygen demand was negligible. Dredging in all seasons could effectively control phosphorous release from the sediment at both sites; however, ammonium release might pose a risk to the water body for a short period after dredging. The seasons with low temperature (non-growing season) are considered the most suitable time for dredging engineering projects, because of the low concentrations of nitrogen and phosphorus in the porewater. To increase the efficiency of internal loading control, greater attention should be paid to determining the optimum dredging depth in the decision-making process. The long-term effectiveness of dredging time on the control of internal loading needs further study.

Supplementary Materials: The following are available online at http://www.mdpi.com/2073-4441/10/11/1606/s1, Figure S1: Schematic diagram of fabrication process for the dredged and undredged sediment cores in the microcosm experiment; Figure S2: Schematic and physical photograph of Peeper.

Author Contributions: J.-C.Z. and J.-H.Y. designed the experiments; J.-H.Y. and X.-L.Z. performed the experiment; S.-L.W. and D.-H.L. analyzed the data; C.-X.F. provided laboratory equipment; J.-C.Z. wrote the article.

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Abbreviations

The following abbreviations are used in this manuscript:

TP	Total phosphorus
TN	Total nitrogen
OC	Organic carbon
LOI	Loss on ignition
Fe	Iron
SRP	Soluble reactive phosphorus
SOD	Sediment oxygen demand

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