Algal Accumulation Decreases Sediment Nitrogen Removal by Uncoupling Nitrification-Denitrification in Shallow Eutrophic Lakes

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ABSTRACT: In eutrophic lakes, the decay of settled algal biomass generates organic carbon and consumes oxygen, favoring sediment nitrogen loss via denitrification. However, persistent winds can cause algae to accumulate into dense mats, with uncertain impacts on sediment nitrogen removal. In this study, we investigated the effects of algal accumulation on sediment nitrogen removal in a shallow and eutrophic Chinese lake, Taihu. We found that experimental treatments of increased algal accumulation were associated with decreased sediment nitrogen losses, indicating the potential for a break in coupled nitrification-denitrification. Likewise, field measurements indicated similar decreases in sediment nitrogen losses when algal accumulation occurred. It is possibly caused by the decay of excess algal biomass, which likely depleted dissolved oxygen, and could have inhibited nitrification and thereby denitrification in sediments. We estimate that if such algal accumulations occurred over 20% or 10% of lake area in Taihu, sediment nitrogen removal rates decreased from 835.6 to 167.2 and 77.2 μmol N m⁻²h⁻¹, respectively, during algal accumulation period. While nitrogen removal may recover later, the apparent nitrogen removal decrease may create a window for algal proliferation and intensification. This study advances our knowledge on the impacts of algal blooms on nitrogen removal in shallow eutrophic lakes.

INTRODUCTION

Nitrogen plays an essential role in the balance and stability of lake ecosystems. Excessive nitrogen can cause eutrophication, reduce biodiversity, and degrade water quality. Denitrification is an important process in the nitrogen cycle, converting nitrate to nitrogen gas and hence permanently removing nitrogen from lake ecosystems. It has been estimated that 43% of nitrogen loads are removed by denitrification in river networks before being discharged to Taihu in China.

In aquatic ecosystems, the processes of denitrification and nitrification often work together to achieve effective nitrogen removal. Nitrification is the biological oxidation of ammonia to nitrate followed by the oxidation of nitrite to nitrate, which is in turn a substrate for denitrification. When coupled nitrification-denitrification is broken, effective nitrogen removal will be limited because denitrifiers will have to rely solely on external “new” nitrate inputs. During algal blooms in eutrophic lakes, the decay of algal biomass supplies organic carbon for heterotrophic denitrifiers. Furthermore, the accumulation of decaying material can physically inhibit oxygen penetration in sediments, creating favorable conditions for denitrification. Chen et al. (2011) demonstrated that the addition of algal biomass accelerated denitrification and enhanced the loss of nitrate from lakes. While relatively small additions of algal biomass may enhance denitrification, the impact of large algal accumulations on denitrification remains unclear.

Wind-driven currents are ubiquitous hydrodynamic features of inland waters, especially shallow lakes. These wind-driven...
Currents occur when wind energy is transferred to the water and dissipated through the generation of relatively high intensity currents.19 Wind-driven currents control the spatial distribution of algae in lentic waters through the advective movement of superficial water masses in a downwind direction.20 Many species of bloom-forming algae are positively buoyant, containing intracellular gas vesicles which help them maintain a position close to the air–water interface. When wind blows in one direction, these positively buoyant algae easily concentrate at the downwind shore.20,21 In the shallow eutrophic Chinese lakes, Taihu and Chaohu, dense algae mats have been frequently observed in the downwind northwestern region during the algal bloom season, particularly following persistent southeast winds.19−21 In such zones with algal accumulation, excessive algal biomass settles on the sediment surface after algae die, and its decay can deplete dissolved oxygen (DO) toward hypoxia or anoxia at the sediment-water interface. These conditions may inhibit sediment nitrification (conventionally performed by obligate aerobes), possibly breaking coupled nitrification-denitrification and limiting sediment nitrogen removal. Moreover, in the newly algae-free zone (from which algae were transported), the decrease of algae derived organic carbon may limit substrates for denitrifiers, thereby potentially slowing down sediment denitrification rates.22 Hence, we hypothesize that the formation of wind-driven aggregations of algae can break coupled nitrification-denitrification via two related mechanisms: (1) hypoxia/anoxia limitation of sediment nitrification in the area of algal accumulation and (2) organic carbon limitation of sediment denitrification in the area from which the algae originated. The combination of these factors may limit net nitrogen removal from sediments during algal bloom seasons in shallow eutrophic lakes.

In this study, sediment nitrogen removal was investigated in Taihu, a shallow and eutrophic lake in China. Monthly nitrogen removal rate in the sediment was monitored for one year in the Meiliang Bay, where algal accumulation often occurs during the summer-time algal bloom season. Incubation experiments under controlled conditions were also conducted to precisely study how algal accumulation affected sediment nitrification and denitrification processes by measuring nitrogen transformation and the associated microbes. The objective of this study was to explore the impacts of algal accumulation on sediment nitrification-nitrification and thereby nitrogen removal in shallow eutrophic lakes, which adds to our understanding of the impacts of algal blooms on nitrogen removal in shallow eutrophic lakes.

## MATERIAL AND METHODS

### Study Area.
Taihu, the third largest freshwater lake by surface area in China, is situated along the lower reaches of the Yangtze River floodplain in the coastal plain of China (Figure 1). It has a surface area of 2338 km², a catchment area of 36500 km², and a mean water retention time of 284 days.22 The lake experiences a subtropical monsoon climate with prevailing winds from the southeast in summer.22 With the rapid agricultural and industrial development of this region in recent decades, excessive nutrient loadings have driven the lake to a hypereutrophic state, causing frequent severe algal blooms in summer every year.23 Our study sites were located in Meiliang Bay, which occupies the northwestern region of Taihu. This bay has a surface area of 123 km² and a mean depth of 1.8 m.24 Driven by southeast winds, algal accumulation often occurs in Meiliang Bay, forming dense algal mats during algal bloom seasons (Figure 1B).20

### Field Surveys and Incubation Experiments.
Monthly nitrogen removal rates in Meiliang Bay sediments were investigated at three sites (S1:31°30′53″ N, 120°10′31″ E; S2:31°30′21″ N, 120°10′28″ E; S3:31°29′37″ N, 120°12′16″ E) by measuring net nitrogen gas fluxes from intact sediment cores. Once per month, intact sediment cores (9 cm in diameter and 50 cm in height) were collected at each site in triplicate using a Jenkin corer in 2016−2017. Nitrogen gas fluxes were measured in situ according to the description below. DO in the water was measured using a multisensor probe (YSI 6600, Yellow Springs Instruments, USA); nitrate was measured by a flow injection analyzer (SAN++ System, SKALAR, Netherlands), with an average RSD of < 5%; chlorophyll a (Chl-a) was determined using the hot ethanol method, respectively.25

To precisely explore how algal accumulation affects the coupled nitrification-denitrification in sediments, twenty-seven intact sediment cores were collected for additional incubation experiments under controlled conditions, whereby the variations of oxygen profile, nitrifier and denitrifier abundances, and nitrogen removal rate in the sediment were monitored.
after varying additions of algal biomass. The concentrations of inorganic nitrogen in the overlying water of sediment cores were also measured to add the understanding of sediment nitrification and denitrification processes over the incubation period. Sediment cores were collected at site S3 in Meiliang bay on June 15, 2016, which were collected separately from the monthly samples described above. Each core contained 15 cm of sediments and 30 cm of water column. The physiochemical properties of sediment and water samples in sediment cores are presented in Table S1. Cores were carefully transported to the laboratory in the dark within 4 h of collection and then left overnight at ambient lake temperature (25 °C) in the dark to avoid the disturbances caused by transportation. On the following morning, three cores were randomly selected for initial nitrogen removal rate measurements, which were subsequently sacrificed in order to collect sediment samples for microbial analysis. The remaining cores were randomly assigned to one of four treatments (each n = 6), which are described as follows: (1) no added algal biomass (Chl-a < 8 μg L⁻¹), simulating the zone where algae have left or algal bloom does not occur (no algal bloom, termed “NAB”); (2) run with algal bloom-equivalent biomass (Chl-a = 30 μg L⁻¹), which refers to the zone where algal bloom occurs before being aggregated to a higher density (algal bloom, termed “AB”); (3) run with 5 times algal bloom-equivalent biomass (Chl-a = 150 μg L⁻¹), which refers to the zone where light algal accumulation occurs (light algal accumulation, termed “LAA”); and (4) run with 10 times algal bloom-equivalent biomass (Chl-a = 300 μg L⁻¹), which refers to the zone where heavy algal accumulation occurs (heavy algal accumulation, termed “HAA”). The amount of algal biomass in each treatment was adjusted by either filtering using a phytoplankton net (64 μm) in the NAB treatment or by adding fresh algal biomass (AB, LAA, and HAA treatments). This algal material was collected using the phytoplankton net at the same location as sediment cores. When algal accumulation occurs, light hardly reaches bed sediment under dense algal canopies. Incubation experiments under dark conditions were conducted indoors at room temperature (24 ± 2 °C) over a period of 30 days.

On day 0 (before the prepared cores were run) and subsequent days 1, 2, 3, 4, 7, 10, 12, 15, 18, 21, 24, 27, and 30, oxygen profiles in sediments were measured using an oxygen microelectrode according to the description below. Water samples (10 mL) were carefully taken 5 cm above the sediment for dissolved inorganic nitrogen analysis (ammonium, nitrate and nitrite) using a 25 mL syringe. On day 10, three cylinders in each treatment were randomly sacrificed to measure nitrogen removal rate and collect sediment samples for microbial analysis. The results of these chemical analyses are presented as mean values. Water removed during sampling was replaced with filtered (Whatman GF/C) surface water collected at the same location as the cores. Dissolved inorganic nitrogen was measured by the flow injection analyzer mentioned above. Dissolved ammonium, nitrate, and nitrite of water used for replacement (Table S1) were used to calibrate dissolved inorganic nitrogen in the cylinders.

**Nitrogen Removal Rate Measurement.** Nitrogen removal rate was analyzed by measuring net nitrogen gas fluxes using intact sediment cores (Figure S1) according to Neiss et al. (2012), which was calculated based on the linear rate of nitrogen gas accumulation as a function of time. Before sample collection, the system was carefully sealed with a gastight mobile rubber without air bubbles in it and left to stand for 10 min to ensure equilibrium according to our preliminary experiments. Water samples were collected every 5 min over a 20 min period and stored into a 12 mL pre-evacuated Exetainer vial (839 W, Labco, UK) after adding 0.2 μL of saturated HgCl₂ solution. The concentration of dissolved nitrogen gas in the water was measured using a membrane inlet mass spectrometry (MIMS) system (Bay Instruments, Easton, MD, USA). In the field, nitrogen removal rate was immediately measured *in situ* after sediment core collection. Anammox is another contributor to nitrogen removal in lake sediment, but it is likely slow relative to denitrification in Taihu. We consider these net nitrogen gas flux measurements to be representative of potential denitrification rates.

**Sediment Oxygen Profile Measurement.** Sediment oxygen profile was determined by introducing an oxygen microelectrode (OX-50, Unisense, Denmark) into the sediment at 200-μm intervals according to de la Rosa and Yu (2005). The microelectrode has a tip diameter of ~50 μm, which is held by a Unisense MM33-2 micromanipulator and connected to a Unisense PA2000 picameter coupled with a computer data-acquisition system (Figure S2). During each measurement, the microelectrode was allowed to equilibrate for 5 s at each depth interval before measuring for 3 s. Prior to use on each sampling day, the microelectrode was calibrated by a 2-point calibration using air aerated water for 100% saturation and 0.1 M ascorbic acid solution for 0% saturation. Oxygen penetration depth refers to the depth from the sediment-water interface to the first point in the tail of oxygen profile where the oxygen concentration remains consistently at the 0 mg L⁻¹. The slope of oxygen profile steadily increased toward a maximum within the first millimeter below the initial concentration decrease, and the position of this maximum gradient was assigned as the sediment-water interface according to Rabouille et al. (2003).

**Microbial Abundance Analysis.** Nitrifiers and denitrifiers in sediments were quantified using the quantitative polymerase chain reaction (qPCR) method. There are many functioning genes involved in nitrification and denitrification processes, of which amoA, nirS, and nosZ were chosen as gene markers in this study. The amoA is a gene marker of ammonia-oxidizing bacteria, which are sensitive indicators of nitrification and play a more important role in the nitrification in Taihu Lake; the nirS and nosZ are commonly used gene markers of denitrifiers, which encode nitric oxide reductase and nitrous oxide reductase, respectively. Sediment samples were stored at ~80 °C for further DNA extraction using a FastDNA PowerMax Soil DNA Isolation Kit (MP Biomedical, USA) according to the manufacturer’s instructions. This DNA subsequently served as a template for qPCR amplification. The qPCR assay was performed using the primer amoA1F/amoA2R targeting nitrifier amoA gene, the primer cd3αF/R3 cd targeting denitrifier nirS gene, and the primer nosZ2F/nosZ2R targeting denitrifier nosZ gene, respectively. Gene copies were amplified and quantified in a Bio-Rad cycler (CFX Manager, Bio-Rad, Hercules, CA) equipped with the iQ5 real-time fluorescence detection system and software (version 2.0, Bio-Rad, USA). All reactions were completed in a total volume of 20 μL containing 10 μL of SYBR Premix Ex Taq™TM (Toyobo, Japan), 0.5 mM of each primer, 0.8 μL of bovine serum albumin (3 mg mL⁻¹, Sigma, USA), double distilled water, and template DNA. The qPCR program for amoA was as follows: 95 °C for 60 s followed by 40 cycles of 95 °C for 30 s and 57
5°C for 45 s and 72°C for 60 s. The qPCR program for nirS and nosZ commenced with 95°C for 60 s, followed by 40 cycles of 95°C for 30 s and 57/60°C (nirS/nosZ) for 45 s and 72°C for 60 s. A standard curve was established by serial dilution (10^-2 to 10^-8) of known concentration plasmid DNA with the target fragment. All PCRs were run in triplicate on 96-well plates (Bio-Rad, USA) sealed with optical-quality sealing tape (Bio-Rad, USA). Three negative controls without the DNA template were included for each PCR run. The amplification efficiency of 90%-100% and R^2 of > 0.99 for each qPCR run were accepted for further calculation.

Statistical Analysis. One-way analysis of variance (ANOVA) was used to test the statistical significance of differences between treatments. Posthoc multiple comparisons of treatment means were performed using the Tukey’s least significant difference procedure. Prior to analysis of variance, the normality test using Kolmogorov-Smirnov and variance homogeneity test were conducted. Logarithmic transformation was also conducted and the transformed data were examined to ensure satisfaction of ANOVA assumptions. All statistical calculations were performed using SPSS 22.0 (SPSS Inc., North Chicago, IL, USA), and the level of significance was P < 0.05 for all tests.

RESULTS

Sediment Oxygen Profiles. Over the 30-day incubation period, oxygen penetration depth in sediments was relatively consistent at 7.6–9.0 mm in the NAB treatment but changed greatly in the treatments with added algal biomass throughout the incubation experiments. In the AB treatment, oxygen penetration depth sharply decreased from 8.2 to 0.4 mm on day 1 and immediately began to recover. Severe anoxia (DO ≈ 0 mg L^-1) occurred in treatments with excessive algal biomass, and the length of this hypoxia was greater with increased algal accumulation. In the presence of excessive algal biomass (treatments LAA and HAA), oxygen penetration depth quickly decreased to 0 mm; the oxygen penetration depth began to recover on days 10 and 15 in the LAA and HAA treatment, respectively (Figure 2).

Water Ammonium, Nitrate, and Nitrite. In the NAB treatment, dissolved inorganic nitrogen in the overlying water was relatively stable throughout the experiments, despite slight increases in nitrate and nitrite concentrations early on. The concentrations of ammonium, nitrate, and nitrite were 0.06–0.31, 0.08–0.67, and 0–0.02 mg L^-1, respectively. In the treatments with added (AB) or aggregated algal biomass (LAA and HAA), ammonium accumulation and nitrate increase were observed within initial incubation days, with ammonium accumulation becoming larger with more algal biomass. Ammonium concentration reached a maximum of 1.55 mg L^-1 and lasted for 7 days in the AB treatment. Similarly, ammonium reached a maximum of 2.01 mg L^-1 and lasted 15 days in the LAA treatment, and reached a maximum of 2.17 mg L^-1 and lasted 24 days in the HAA treatment. Moreover, in the presence of excessive algal biomass, there were notable increases in nitrate and nitrite at the end of incubation experiments. On day 30, nitrate and nitrite increased to 1.67 and 0.02 mg L^-1 in the LAA treatment, and 1.73 and 0.09 mg L^-1 in the HAA treatment, a factor of 9.8 rise relative to the average surface water concentration.

Nitrogen Removal Rate. The nitrogen removal rate was 66.4 µmol N m^-2 h^-1 in the NAB treatment and increased to 835.6 µmol N m^-2 h^-1 in the AB treatment. However, nitrogen removal rate decreased when algal biomass was excessive. Nitrogen removal rate was only 570.3 µmol N m^-2 h^-1 in the LAA and even lower in the HAA treatment, at 173.9 µmol N m^-2 h^-1.

Nitrifier and Denitrifier Abundance. Compared with the control NAB treatment (amoA gene = 1.61 × 10^6 copies g^-1 sediment), nitrifier abundance was relatively high in the AB treatment and relatively low with dense algal aggregations (LAA and HAA). The amoA gene was 2.28 × 10^6, 0.46 × 10^6, and 0.99 × 10^6 copies g^-1 sediment in the AB, LAA, and HAA treatments, respectively. Unlike nitrifiers, denitrifier abundance in treatments with algal biomass (nirS: 2.21 ± 0.51 × 10^8; nosZ: 0.92 ± 0.18 × 10^8) was significantly greater (P < 0.05) than those with no algal bloom (nirS: 0.85 ± 0.39 × 10^8; nosZ: 0.51 ± 0.15 × 10^8). However, denitrifier abundance in the treatments with dense algal aggregations (LAA and HAA) was lower than the treatment with algal bloom-equivalent biomass additions (AB). The nirS and nosZ gene reached 0.85 × 10^9 and 0.51 × 10^9 copies g^-1 sediment in the NAB treatment, 2.69

Figure 2. Changes of oxygen penetration depth in sediments in incubation experiments. (A) NAB; (B) AB; (C) LAA; (D) HAA.
processes in lake sediments. Specifically, Shi et al. found that redox-sensitive methanogens/methano-
genera declined during algal bloom seasons, which is concerning in the restoration of eutrophic lakes.

Monthly Sediment Nitrogen Removal Rate and Water Characteristics in Meiliang Bay, Taihu. In Meiliang Bay, annual nitrogen removal in sediment was estimated to 1.6 × 10^9 and 1.15 × 10^9 copies g^−1 sediment in the AB treatment, 1.50 × 10^9 and 0.71 × 10^9 copies g^−1 sediment in the LAA treatment, and 2.44 × 10^9 and 0.90 × 10^9 copies g^−1 sediment in the HAA treatment, respectively.

In shallow lakes, surface sediments are often oxic due to oxygen penetration from oxygen-rich overlying water with rapid air−water gas exchanges. In the present study, we observed deep oxygen penetration in sediments for treatments without added algal bloom material (Figure 2). Following algal biomass additions, rapid algal decay consumed oxygen and decreased the oxygen penetration depth into sediments, favoring sediment denitrification for nitrogen removal. Accordingly, a higher abundance of sediment denitrifiers and rapid nitrogen removal were detected in treatments with added algal bloom biomass when compared with those in the control treatment without algae. In the presence of excessive algal biomass, the decay of algal biomass created favorable conditions for denitrification (replete organic carbon, anoxia). However, these anoxic conditions may have also inhibited nitrification, potentially limiting the supply of nitrate for subsequent denitrification. This may partially explain the low nitrogen removal rate found in the HAA treatment, relative to LAA and AB (Figure 4).

In aquatic systems, the synergy of nitrification and denitrification achieves effective nitrogen removal. Nitrification converts ammonium to nitrate in oxic surface sediments, supplying substrates for nitrogen removal by denitrification in anoxic subsurface sediments. When algal blooms occur in shallow eutrophic lakes, the decay of settled algal biomass supplies organic carbon and decreases oxygen penetration depth into sediments, favoring sediment denitrification for nitrogen removal. Accordingly, a higher abundance of sediment denitrifiers and rapid nitrogen removal were detected in treatments with added algal bloom biomass when compared with those in the control treatment without algae.

DISCUSSION

In shallow lakes, surface sediments are often oxic due to oxygen penetration from oxygen-rich overlying water with rapid air−water gas exchanges. In the present study, we observed deep oxygen penetration in sediments for treatments without added algal bloom material (Figure 2). Following algal biomass additions, rapid algal decay consumed oxygen and decreased the oxygen penetration depth in sediments (Figure 2). In our treatments with dense algal accumulations, the decay of excessive algal biomass depleted dissolved oxygen, causing severe anoxia in sediments (Figure 2CD). As algal biomass was gradually consumed, algae-induced anoxia slowly receded with oxygen delivery from the air. However, the duration of this anoxia varied with the amount of algal biomass added. In the presence of excess algal biomass (LAA, HAA), anoxia persisted for a longer time (Figure 2C). In fact, this anoxic duration nearly doubled when algal biomass was doubled (Figure 2CD). This redox variation will affect redox-sensitive microbes and can alter biogeochemical processes in lake sediments. For instance, Shi et al. (2018) found that redox-sensitive methanogens/methano-
trophs were inhibited in sediments when anoxia/hypoxia was reduced, causing a decrease in methane emission from a eutrophic lake. See Figures 3 and 4.
summer algal bloom and accumulation period. At the same time, wind-driven algal accumulations also generate large areas of relatively low algal biomass, where decreased organic carbon supplies and strong oxygen penetration may also hamper denitrification. In support of this hypothesis, we found relatively low nitrogen removal rates in the control treatment without added algal biomass (Figure 4). Nitrates from anthropogenic sources and the zone without algal accumulation may be transported to the zone with algal accumulation and subsequently support its denitrification. However, nitrates in the overlying water maintained extremely low levels, and nitrogen removal recovery was not detected during the algal accumulation period in our field surveys (Figure 6), indicating the limited contribution of external nitrate inputs during the severe algal accumulation. During algal blooms in Taihu, if algae are aggregated by wind to a region that is 20% of the total lake surface area, 80% of the lake area will be relatively poor in algal biomass. In this case, nitrogen removal rate is estimated to decrease from 835.6 to 167.2 μmol N m⁻² h⁻¹ by 80.0% according to nitrogen removal rates in our simulation experiments; if algal biomass accumulates to 10% of lake area, creating 90% of lake area poor in algal biomass, the nitrogen removal rate is estimated to further decrease to 77.2 μmol N m⁻² h⁻¹ by 90.8% (Figure S3). The measured nitrogen gas fluxes in the field (Figure 6) were generally much less than those in the simulation experiments (Figure 4). It is plausible that nitrogen gas fixation could have been greater in the field but inhibited in the cores, causing nitrogen gas fluxes in the algal accumulation treatments to be relatively high. Moreover, while our study focused on microbial abundance, it is likely that the activity of nitrifiers and denitrifiers may have also changed under the shift of redox conditions caused by algal biomass, further altering nitrogen removal rate. This deserves further study using other molecular biology techniques, such as DNA/RNA-based stable isotope probing. In fields, hypoxia/anoxia was not detected in the water during algal accumulation (Figure 6B), since dead algae cells often settle on sediment surface and decay. However, denitrification possibly occurs inside suspended particles, contributing to nitrogen removal, which needs further studies to quantify the contribution.

As anoxia receded (Figure 2), a decrease in ammonium (Figure 3A) and subsequent increases in nitrate (Figure 3B) and nitrite (Figure 3C) were observed at the end of incubation experiments in the treatments with dense algal aggregations. We suggest that this was caused by an “awakening” of the nitrifying community, helping to restart coupled nitrification-denitrification. In our monthly field measurements, we detected a gradual increase in nitrogen removal rate post algal accumulation (Figure 6). The negative impacts of algal accumulation on nitrogen removal can weaken or disappear later, as seen in the increased nitrogen gas flux in the fall months in Meiliang Bay (Figure 6); and the decreased nitrogen loss in the algal accumulation period may be just the redistribution of nitrogen removal amount over the whole year. However, we suggest that frequent algal accumulations in shallow lakes may cause considerable negative impacts on nitrogen removal and hence lake restoration. The decreased nitrogen removal can make more nutrients available for algal proliferation, thereby intensifying existing algal blooms. As the saying goes, “United we stand; divided we fall”. During blooms, it appears that algae united by wind-driven currents can generate conditions favorable for nutrient retention and subsequent proliferation. On the contrary, algal blooms that are evenly distributed during quiescent conditions may create denitrification-favorable conditions, encouraging the loss of nutrients vital for further bloom development. This may be a new law of algal survival in shallow eutrophic lakes.

Figure 5. Average nitrifier and denitrifier abundance in each experimental treatment. (A) amoA gene; (B) nirS gene; (C) nosZ gene. Error bars indicate standard deviations.

Figure 6. Sediment nitrogen removal rate and water characteristics in Meiliang Bay, Taihu. (A) Sediment nitrogen removal rate and water Chl-a; (B) water DO and nitrate. Annual nitrogen removal was estimated by the trapezoidal method using MATLAB (MathWorks, USA). The data were collected in 2016–2017. Error bars indicate standard deviations.
Experimental designs for nitrogen gas flux and sediment oxygen profile analyses, estimations of nitrogen removal reduction on the lake scale, and physiochemical properties of sediment cores (PDF)

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Notes
The authors declare no competing financial interest.

ACKNOWLEDGMENTS

This study was supported by the National Natural Science Foundation of China (No. 41621002, 41701112, 51709181, 51979171), Starting Research Fund of Nanjing University of Information Engineering (No. 20181015).

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