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Effects of floating-leaved macrophytes on water quality and phytoplankton: an *in situ* experiment in a Chinese shallow lake

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ABSTRACT

In this study, *in situ* experiments were conducted to evaluate the effects of floating-leaved macrophytes (FLMs, i.e. *Trapa uncinata* and *Nymphaea tetragona*) on water quality and algal growth and to investigate the potential inhibition of phytoplankton growth by FLMs. Enclosures were established in a shallow subtropical eutrophic lake, and FLMs were planted therein. Open water areas in the enclosure with FLMs, an enclosure without FLMs, and the lake water surrounding the enclosures served as the control groups. Physical parameters, nutrient contents, and algae were evaluated for six sampling sites at 10-d intervals. Treatment with FLMs significantly promoted water transparency and reduced turbidity. However, the concentrations of nutrients were similar in the different sites, despite the nutrient absorption by FLMs. The overall algal abundance was reduced in the presence of FLMs, and this was shown in chlorophyll-a, algal density, and biomass. FLM treatments promoted changes in the algal community composition, benefitting metaphytic species adapted to higher transparency. A principal components analysis showed that algal attributes were higher in the control areas than in the FLM-covered areas, despite the similar nutrient concentrations. This indicated that the inhibition of algae in this study was based not on nutrient competition but instead on the shading effect. Using FLMs as a management tool to create a clear water state may be one solution for turbid waters in subtropical eutrophic shallow lakes.

Keywords: Floating-leaved macrophytes; Shallow lake; Shading effect; Phytoplankton

1. Introduction

Shallow lakes have two alternative stable water states: a macrophyte-dominated clear state and a phytoplankton-dominated turbid state [1,2]. The ability and application of submerged macrophytes to control lake eutrophication has been studied intensively in recent years [3–6]. These studies confirmed the critical role of macrophytes in structuring food web dynamics and the equilibrium of the ecosystem within lakes. The main mechanisms by which submerged macrophytes maintain a clear water state are thought to be nutrient competition with phytoplankton, providing refuge for zooplankton, producing allelopathic substances and inhibiting sediment resuspension [7,8].

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Many lake managers have tried to re-establish or enhance macrophyte abundance to restore eutrophic waters, with the intention of improving water quality, increasing water transparency, and reducing phytoplankton biomass [9].

Lakes in the middle and lower reaches of the Yangtze River Estuary have suffered from algal blooms for several decades [10–13]. There have been many attempts to control harmful algae through restoration of submerged macrophytes [14,15]. However, most restoration experiments have been conducted in the littoral zone at shallow depths, where submerged macrophytes can easily survive. In deeper, open water (OA), with depths greater than *ca.* 2 m, submerged macrophytes have low survival, primarily due to the low transparency.

Dianshan Lake is a shallow lake in the Yangtze River Estuary. Before the 1950s, the lake was covered by aquatic macrophytes, the water was transparent enough for the bottom to be visible, and the water column was mesotrophic [16]. The development of aquaculture in the lake and the discharge of wastewater to the lake from industry and agriculture since the 1980s led to a eutrophic state after 2000 and a decline in the macrophyte population. Algal blooms have occurred every summer since 2000. The lake water is now eutrophic, with a Secchi depth of less than 0.6 m. Eutrophication resulted in negative effects on the ecological health and economic utilization of the lake [17–19]. A need for water restoration has been perceived by scientists, local authorities, and the public.

Floating-leaved macrophytes (FLMs) can grow well in eutrophic shallow lakes. They can also inhibit algal growth via the following mechanisms. First, they can assimilate nutrients from the water body and the sediment and compete with algae for nutrient absorption. It has been reported that P stocks in FLMs reached high proportions (16.80–156.25%) of the annual net flux of P in Lake Grand-Lieu, France [20]. Second, there might be allelopathic effects of FLMs on algae. Nemoto et al. found a significant negative correlation between the concentrations of chlorophyll a and the growth areas of *Trapa* spp. in Lake Sagata, Japan [21]. They considered that the antagonistic relations could be explained based on a likely scenario of allelopathic effects on the development of cyanobacteria by *Trapa* spp. vegetation. Many studies also proved the allelopathic effects caused by free-floating macrophytes (FFMs) such as water hyacinth [22,23]. Third, as with FFMs, a shading effect may also be expected to inhibit algal growth [24]. Other factors such as reduction of resuspension and internal P loading could also contribute to the inhibition of algal growth [25].

Like submerged macrophytes, FLMs play an important role in aquatic ecosystems and can also create a clear water state [20]. Many studies have focused on the interactions between FFMs and phytoplankton. However, to our knowledge, few studies have focused on the use of FLMs for controlling undesired algal growth. Thus, the goals of this paper were (a) to evaluate the effects of FLMs (*Trapa uncinata* and *Nymphaea tetragona*) on water quality and algal growth through a field study and (b) to investigate a method for creating a clear water state in eutrophic shallow lakes by planting FLMs.

2. Materials and methods

2.1. Study site

Dianshan Lake (120˚58´, 31˚07´) is located in the lower reaches of the Yangtze River, China (Fig. 1). It is a shallow, subtropical, freshwater lake and the main water source for the city of Shanghai. The lake has an area of 63.7 km^2, with average and maximum depths of 2.1 and 3.6 m, respectively. It receives water from Taihu Lake and discharges into the East China Sea via rivers. There is a slow current of 0.02 m s^{-1} within the lake, flowing from the northwest to southeast. The water residence time of the lake is 29 d. The lake supports multiple functions and activities, including supplying municipal water, recreation, fishing, irrigation, flood storage, and waterborne transportation.

2.2. FLM cultivation

A large experimental enclosure (100 × 200 m) and two small enclosures (20 × 20 m) were established in Dianshan Lake in August 2008 using bamboo, netting, and waterproof fabric (Fig. 1). The sediment was not covered. Seeds of *T. uncinata* were sown (thrown evenly into the water, *ca.* 40 g m^{-2}) in two strips in the large enclosure in January 2010, and they sprouted in early April of that year. Subterraneous stems of *N. tetragona* were transplanted into a small enclosure in April 2009. The macrophytes grew from April through the autumn and began to decline at the end of September. Thus, the study was divided into two time periods: the warm season (9 July–19 September) and the cold season (after 19 September). Once they had died, the macrophytes were removed from the water (Fig. 2).

2.3. Sampling and analysis

Six sampling sites were established: sites T1 and T2 were in the areas covered by *T. uncinata*; site OA was in the open area inside the large enclosure; site

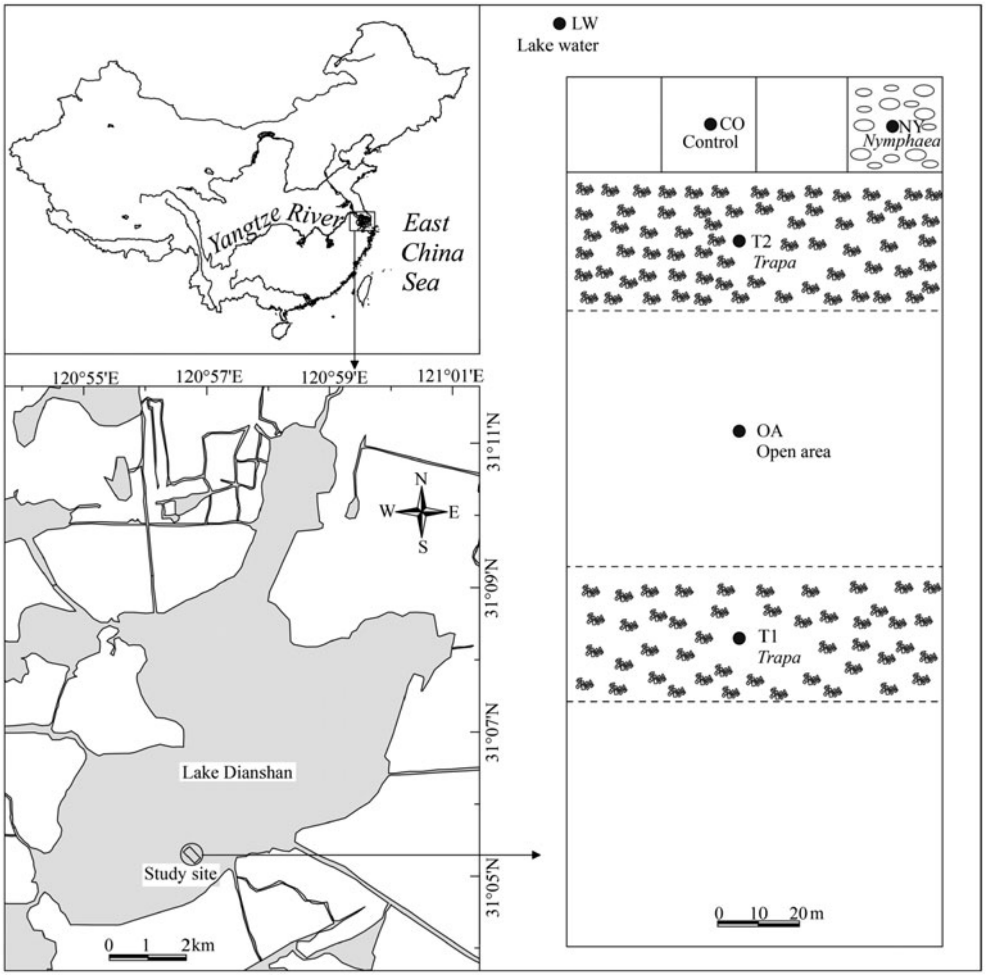

Fig. 1. Study location and diagram of the sampling sites. Continuous lines indicate walls made of waterproof fabric and bamboo, which can separate the water in each enclosure. Dashed lines indicate nets fixed by bamboo.

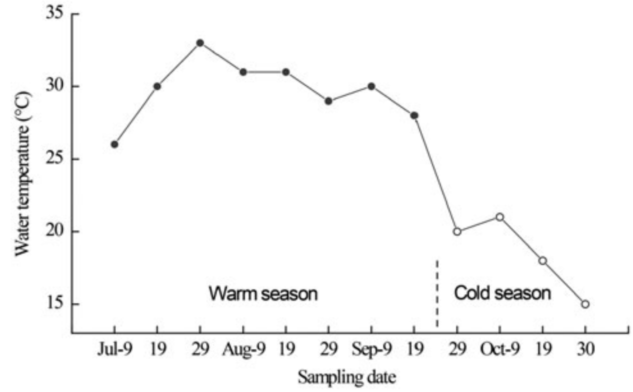

Fig. 2. Water temperatures during the study period.

NY was in a small enclosure with *N. tetragona*; site CO was in a small enclosure without macrophytes; and site LW was outside the enclosures, i.e. in the lake water (Fig. 1). Sites OA, CO, and LW served as the control groups.

The large enclosure was established because we originally wanted to test the wave reduction ability of FLMs and to perform other experiments in that enclosure in future. Because of the large scale and associated costs, no parallel replicates were conducted. However, we conducted time series replicates to compensate for the absence of parallel replicates. T1, T2, and OA were not isolated by waterproof fabric because we wanted to test whether the transparency in OA would be improved enough to restore submerged macrophytes after nutrients were absorbed by FLMs in T1 and T2.

Samples were taken at 10-d intervals at each site from 9 July to 30 October 2010 (except 19 October). Though frequent plankton sampling is desirable because of normal temporal variability and the migratory character of the plankton community, because of limits on our equipment, personnel availability, and time for sample analysis, we selected a 10-d sampling gap. This interval may be similar to the plankton community turnover time. An integrated sample was collected with a 5-L perspex sampler at 0.5 m intervals from the bottom to the top of the water column in each site, and it was then mixed in a plastic bucket. A 1-L portion of the sampled water was preserved for phytoplankton enumeration, and another 1-L portion of water was collected for nutrient analyses [26–28]. Nutrient contents were analyzed according to standard methods [27,29]; the nutrients measured included ammonia nitrogen (NH_4, Nesslerization method), nitrite nitrogen (NO_2, diazotization method), nitrate nitrogen (NO_3, Cd reduction, and diazotization method), total nitrogen (TN, persulfate digestion, and ultraviolet spectrophotometry), soluble reactive phosphorus (SRP, ammonium molybdate method), total phosphorus (TP, persulfate digestion and ammonium molybdate method), chemical oxygen demand with permanganate consumption (COD_{Mn}, acidic potassium permanganate method), and suspended matter (SS). Water transparency (SD) was measured *in situ* using a Secchi disk. Light intensity was determined using a light meter (ZDS-10W, Xuelian, China). Dissolved oxygen (DO) and water temperature (Tem) were measured at 0.5 m intervals in the water column using a portable dissolved oxygen meter (sensION 6, Hach, USA). Turbidity (Tur) was measured using a turbidity meter (WGZ-3, XINRUI, China)

Chlorophyll-a (chla) was measured using the spectrophotometric method [30] after filtration on Whatman GF-C glass filters and extraction in 8 mL of 90% acetone for 24 h. The extraction procedure followed the standard extraction method of the State EPA of China [27] and the APHA [29]. Chlorophyll-a is the primary photosynthetic pigment of all oxygen-evolving photosynthetic organisms and is present in all algae, and it is widely used as an indicator of the quality of freshwater bodies [31]. Phytoplankton samples (1 L) were fixed in acetic Lugol's solution and preserved with 5% formalin. After sedimentation for a week in a 20-cm-tall container, the top algal-free supernatant was carefully siphoned off using a pipette with a "U" tip, to cause minimal disturbance [32]. The sedimentation time is ~36–48 h according to the Chinese standard methods, but we allowed the samples to settle for a week. The rest of the sample was transferred to an opaque glass container, and the volume was adjusted to 50 mL, whereupon it was stored in the dark at constant temperature (4℃) until needed for analysis. After being mixed completely, 0.1 mL of the concentrated sample was placed in a 0.1-mL counting chamber (Palmer-Maloney counting cell), and phytoplankton were counted using a microscope (Nikon, Japan) at a magnification of 400×. Triplicate chambers were counted. If the counting results between chambers differed by more than 15%, more chambers were counted. This procedure followed the Chinese standard methods for lake eutrophication surveys [27,28], and this method has been widely used in other studies [33–39]. Phytoplankton species were identified according to Hu and Wei [40] and Bellinger and Sigee [32]. Algal biomass (wet weight) was obtained by considering algal cells as equivalent geometric shapes [41]. Conversion to biomass assumes that 1 mm^3 of volume is equivalent to 1 mg of fresh weight biomass [42].

The biomass of *T. uncinata* (above-ground parts) was measured by harvesting the aboveground organs of plants in quadrats (size: 1 × 1 m). The aboveground organs were dried at 60℃ until a constant mass was reached, and then they were weighed. The N and P contents were determined from subsamples of the fine powder of dry plants ground using a centrifugal mill. The phosphorus content (%P) was determined using the vanadomolybdophosphoric acid colorimetric method after digestion with HNO_3 and $HClO_4$, and the nitrogen content (%N) was measured using an automatic chemical analysis system (FOSS Kjeltec 2300) after digestion with H_2SO_4. The nutrient standing stocks in the aboveground structures were derived from the biomass, nutrient content, and area covered by the plant.

A principal components analysis (PCA) was employed to reveal the main patterns among sampling sites. All the measured environmental parameters for the 66 samples were used in the PCA. Redundancy analysis (RDA) was used to elucidate the association between the phytoplankton species and the environmental variables in the warm season. We opted for a linear rather than a unimodal model of ordination because preliminary detrended correspondence analysis showed a short gradient length for the biological data (with a maximum gradient length of 1.577) [43]. The RDA used 29 frequently occurring phytoplankton species (i.e. present in more than 10 samples), 10 environmental variables, and 66 samples. The data used in the RDA were $\log(x + 1)$-transformed to down-weight large values. The statistical significance of ordination axes was tested using a Monte Carlo permutation test (499 unrestricted permutations) to guard against interpretation of spurious axes. RDA was performed using the computer program Canoco for Windows 4.5.

3. Results

During the study period, water temperature was greater than 25℃ before 29 September and lower than 25℃ thereafter (Fig. 2), and the FLMs began to wither after 29 September.

The turbidity and SD showed contrasting patterns among the different sites (Fig. 3). SD differed greatly among the sampling sites ($p < 0.05$). It reached approximately 1.7 m in the areas covered by FLMs, whereas in other areas, it was only 0.6 m on average during the warm season. In the warm season, the lowest turbidity and highest SD were found in the region covered by *T. uncinata*, while the highest turbidity and lowest SD were found in the lake water outside the enclosures. Water covered by *N. tetragona* also had a relatively low turbidity. When the macrophytes withered, the differences between sampling sites almost disappeared. SS was significantly correlated with turbidity. The lower SD in 29 September was a result of strong wind.

There was no distinct difference between sampling sites with regard to the concentrations of nitrogen, except that the concentrations of NO_2 and NO_3 were higher in LW than in the other sites (Fig. 4). However, in the warm season, the lowest average concentrations of NO_2 (0.016 mg L^{-1}), NO_3 (0.15 mg L^{-1}), and TN (1.30 mg L^{-1}) were all in OA. The average TN values were higher in CO and LW than in the other sites. In the cold season, TN soared above 3 L^{-1} mg on October 30th at all sites.

The patterns of SRP and TP were similar to those of NH_4 and TN (Fig. 5). No differences in SRP concentration were found among the six sampling sites. There was no consistent difference among the sites concerning TP, but the average concentration of TP in LW (0.16 mg L^{-1}) and CO (0.13 mg L^{-1}) were slightly higher than in the large enclosures (0.12 mg L^{-1}) or NY (0.11 mg L^{-1}) during warm season.

The concentrations of COD_{Mn} and chla exhibited different patterns in the warm season (Fig. 6). Generally, COD_{Mn} concentration changed synchronously in different sites. However, the chlorophyll-a concentrations in CO, LW, and OA fluctuated sharply and were sometimes much higher than in the other sites, but during the warm season, the chlorophyll-a concentration was more stable in the sites covered by plants, except on 9 July. Thus, the average concentration of chla in CO and OA was twice that in the sites covered by plants. The highest chla was found in OA on 9 July.

The light intensity decreased exponentially with water depth (Fig. 7(a)). In the areas covered by *T. uncinata*, the relative light intensity (light intensity at the water surface = 100%) decreased sharply with increasing water depth. This finding is due to shading by high densities of *T. uncinata*, which covered approximately 95% of the water surface of the areas in which they grew, resulting in light intensity at water depths of 0.5 m that was only 2% of the value at the surface.

The area covered by *N. tetragona* showed a similar profile to the OA. In the middle and bottom of the water column, the light intensity in the *N. tetragona* enclosure (4.1%) was even greater than in the OA, due to the increased space between the leaves of *N. tetragona*, which covered approximately 50% of the water surface. The SD in the *N. tetragona* enclosure was markedly higher than in the OA, so the light intensity was higher there.

DO concentrations decreased with increased water depth. DO in a certain water layer also varied between sites and might have been inversely correlated with

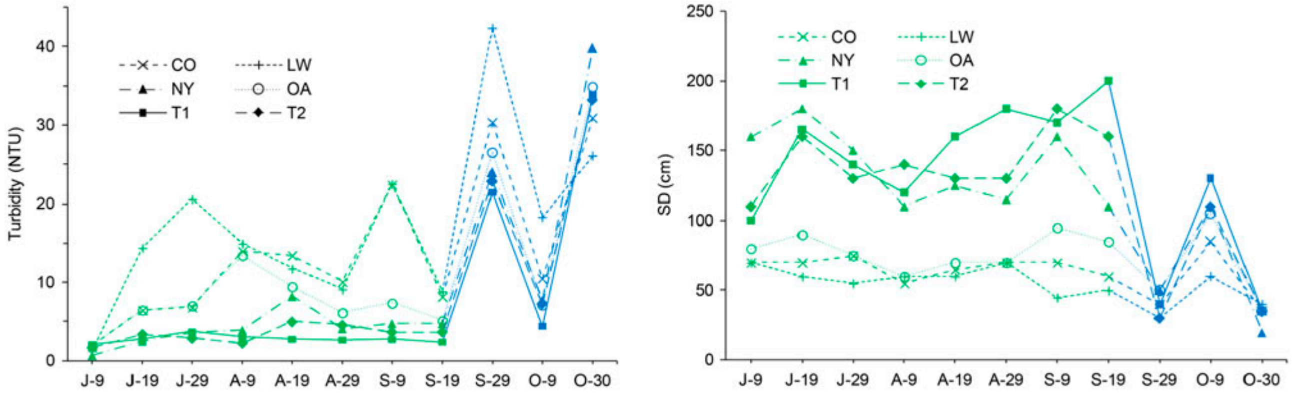

Fig. 3. Dynamics of turbidity (left) and SD (right). The growing season is shown in green, and the cold season is shown in blue.

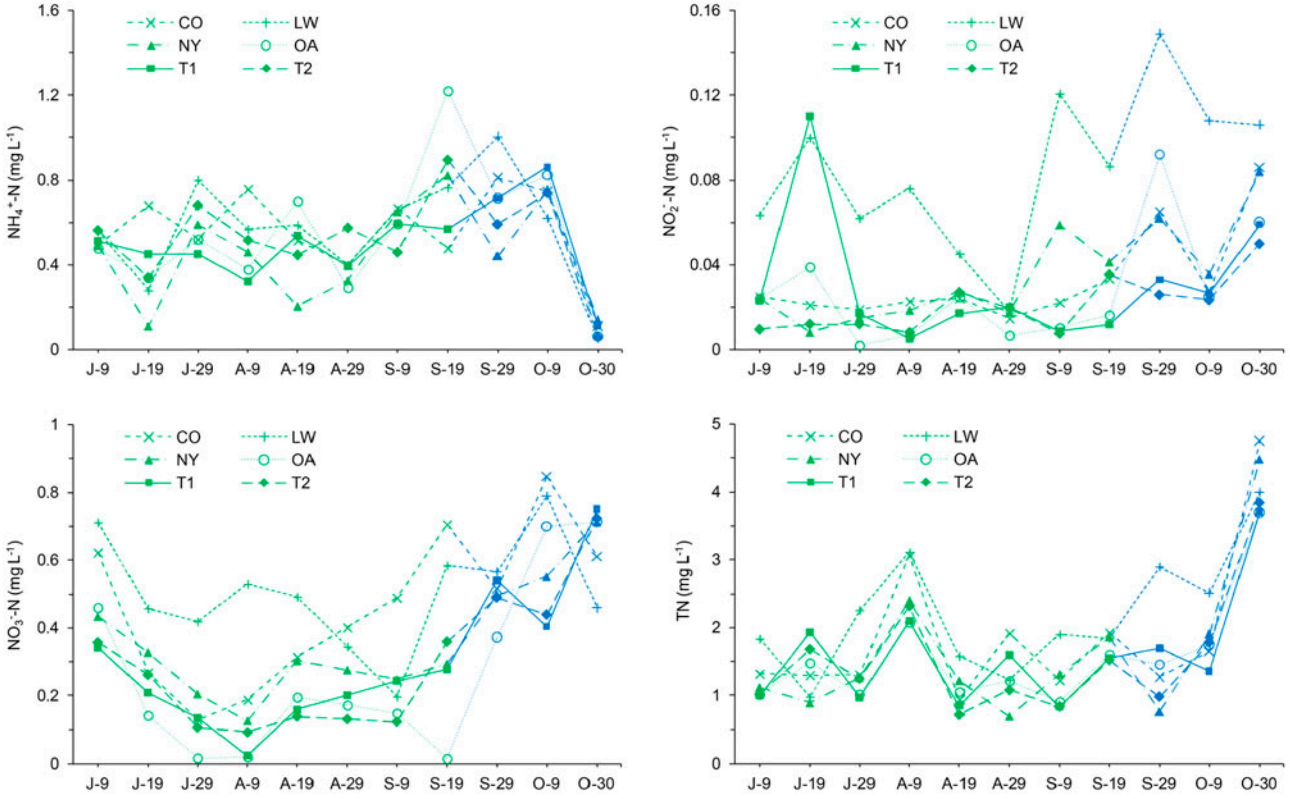

Fig. 4. Inorganic and total nitrogen concentrations for the different sites. The growing season is shown in green, and the cold season is shown in blue.

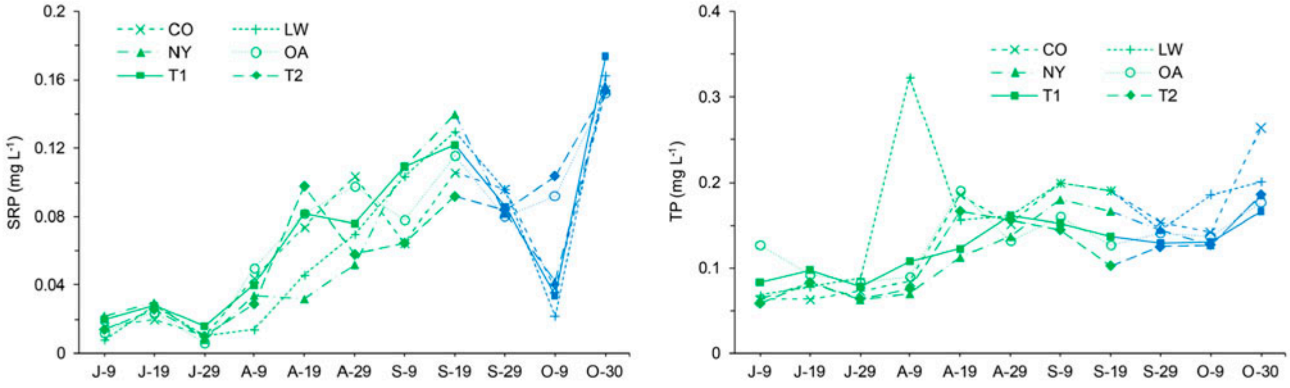

Fig. 5. Phosphorus concentrations at the different sites. The growing season is shown in green, and the cold season is shown in blue.

the coverage of water surface by FLMs in the warm season (Fig. 7(b)). There were no obvious patterns in the cold season.

The density and biomass of algae showed patterns similar to that of the concentration of chla at the different sites (Fig. 8). Both of these measures were more stable in the areas covered by FLMs and highly variable in the OA, resulting in the average algal density and biomass in CO, LW, and OA being much higher than in T2 and NY during the warm season.

A total of 97 algal taxa were recorded in our study. There were 36, 42, 31, 34, 36, and 86 taxa found at sites T1, OA, T2, NY, CO, and LW, respectively. Generally, *Merismopedia glauca* (Ehrenberg) Kützing,

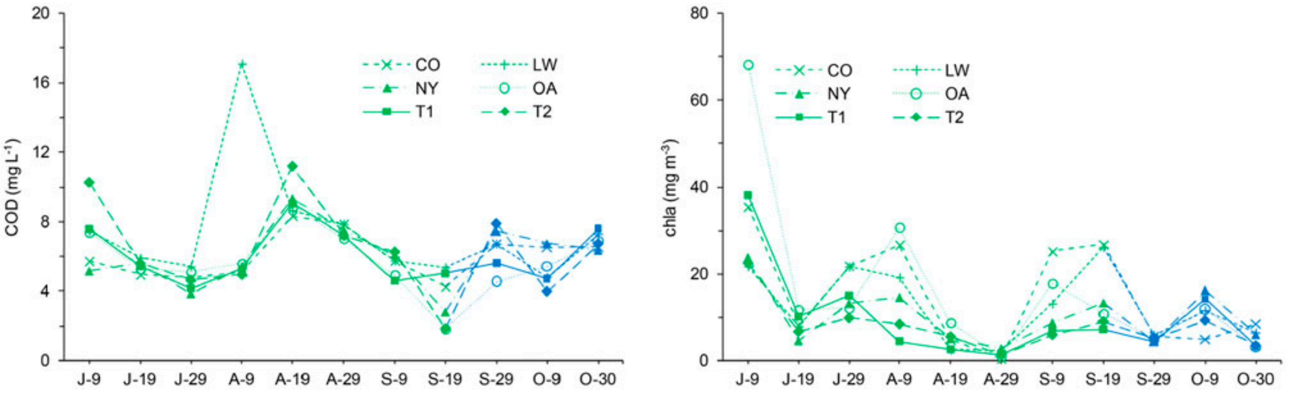

Fig. 6. COD_{Mn} and chla concentrations at the different sites. The growing season is shown in green, and the cold season is shown in blue.

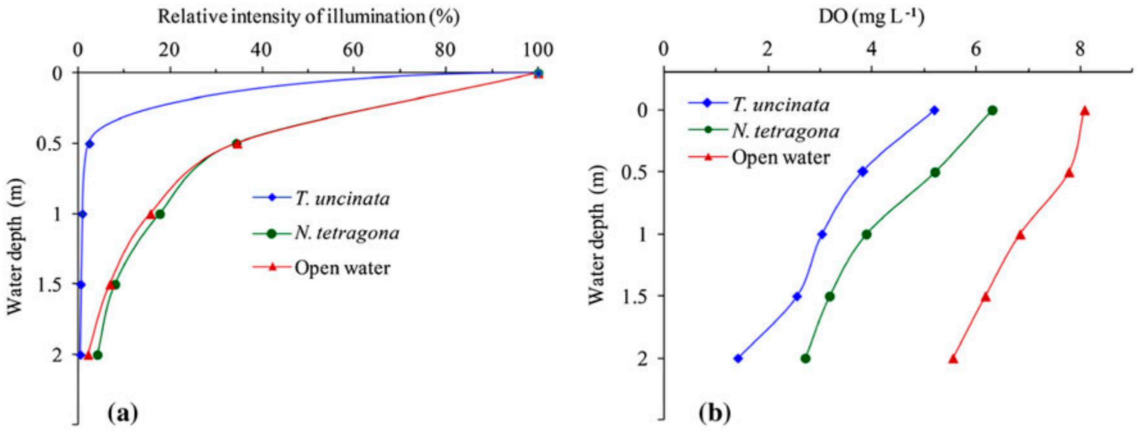

Fig. 7. Profiles of (a) average relative intensity of illumination and (b) DO concentration by water depth during the warm season.

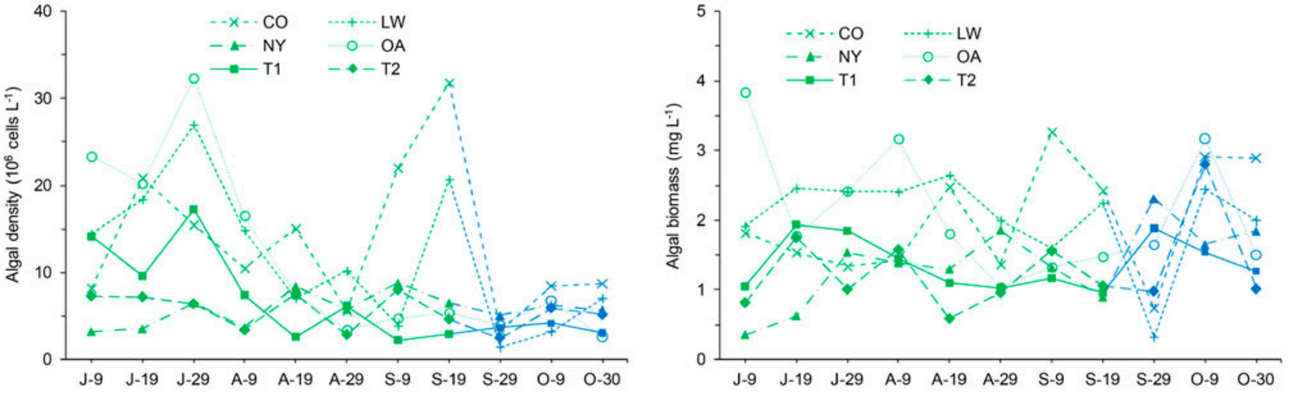

Fig. 8. The density and biomass of phytoplankton at the different sites. The growing season is shown in green, and the cold season is shown in blue.

Phormidium spp. and *Chroococcus minor* (Kützing) Nägeli numerically dominated at most sampling sites. *Chlorella vulgaris* Beyerinck and *Chroomonas acuta* Utermöhl were the most frequent species and were found in all samples. Each of these species accounted for 3–9% of the total phytoplankton density. However, the phytoplankton biomass consisted primarily of *Chlamydomonas* spp., *Synedra* spp. and *Cryptomonas erosa* Ehrenberg.

The PCA was used to examine differences between sampling sites in regard to nutrients and algal growth (Fig. 9). The first principal component (PC1) explained 34.1% of the total variance and was primarily correlated with NO_2 (0.712), NO_3 (0.681), TN (0.772), SRP (0.664), and TP (0.712), i.e. it was indicative of nutrient levels. PC2 explained 18.8% of the total variance and represented algal growth status because it was primarily correlated with chla (0.756), algal density (0.825) and algal biomass (0.769). Most of the samples collected during the warm season from sites covered with FLMs were associated with low nutrient levels and algae growth. However, samples from CO and OA were in the second quadrant, with nutrient levels similar to those of T1, T2, and NY but with more algae. Samples taken during the cold season from all sites showed high nutrient levels.

The RDA revealed three main groups of phytoplankton species as well as three main groups of samples along the SD gradient in the warm season (Fig. 10). The first two axes explained 14.5% of the cumulative variance of the species data. The species–environment correlations of axis 1 and axis 2 were 0.807 and 0.776, respectively. The first four axes explained 22% of the total variance in species data. The Monte Carlo permutation test was significant for the first axis ($F = 4.29$, $p = 0.002$) and for all axes taken together ($F = 1.56$, $p = 0.002$).

Group 1 was characterized by a high SD and consisted of samples taken from the area covered by plants (Fig. 10(b)), algae were represented by more metaphytic species, i.e. *Gomphonema* sp., *Synedra ulna* (Nitzsch) Ehrenberg, *Cymbella* sp., *Tetraedron trigonum* (Nägeli) Hansgirg, *Dactylococcopsis rhaphidioides* f. *falciformis* Printz, *Synechococcus lividus* Copeland, and *Navicula* sp. (Fig. 10(a)). Meanwhile, group 2 mainly contained the samples taken from LW and CO, and it was characterized by a low SD and distinctly separated from the other samples. More cyanophyta such as *M. glauca* (Ehrenberg) Kützing, *C. minor* (Kützing) Nägeli, *Phormidium corium* (C. Agardh) Kützing ex Gomont, as well as *Cyclotella* sp. were found in this group. Other samples, mainly taken from CO and OA, were scattered between groups 1 and 2.

Wave height was approximately 30 cm in the OA but was only 5 cm in the area covered by FLMs, according to our rough measurement. The biomass (wet weight) of *T. uncinata* was 5.54 kg m^{-2} in August, while water content was 85.2% of the wet weight, and the area covered by *T. uncinata* was 8,000 m^2 in total. Nitrogen and phosphorus were 2.12 and 0.27% of the dry weight of *T. uncinata*, respectively. Thus, the nutrient standing stocks in *T. uncinata* were 139 kg of

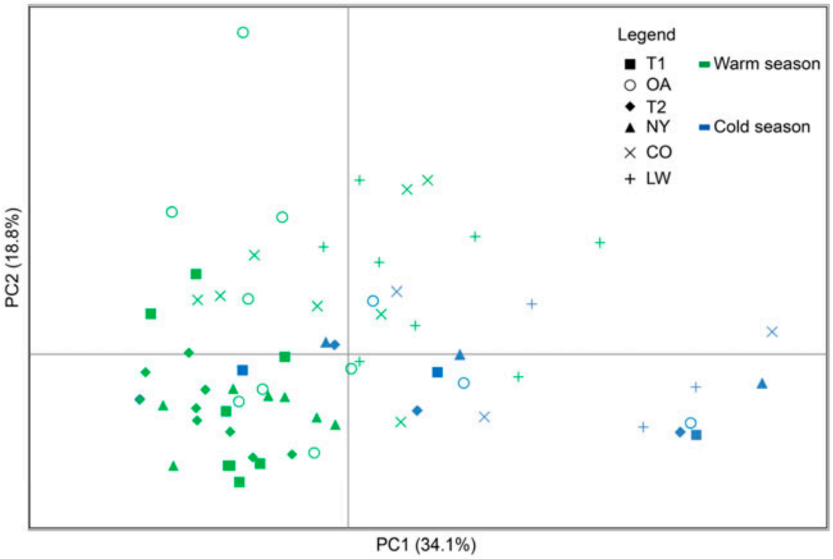

Fig. 9. PCA plot. PC1 primarily correlates with NO_2, NO_3, TN, SRP and TP. PC2 primarily correlates with chla, algal density and algal biomass.

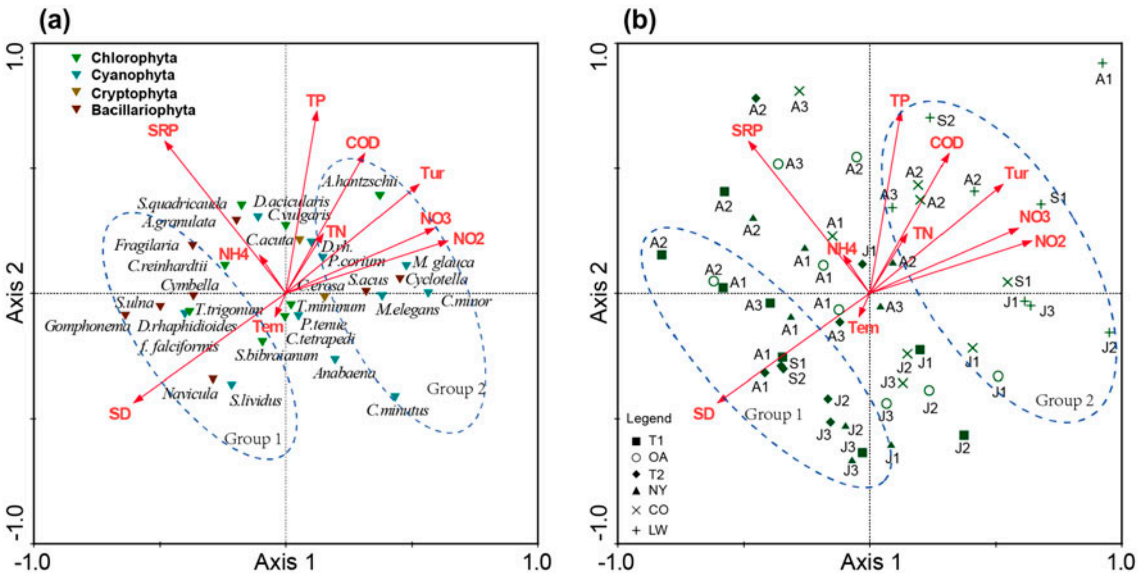

Fig. 10. RDA ordination plots with (a) species and environmental variables and (b) samples and environmental variables. The eigenvalues for the first two axes are 0.104 and 0.046.

nitrogen and 17 kg of phosphorus. Nutrient absorption can theoretically decrease the water nutrient by 0.033 mg L^{-1} d^{-1} of nitrogen and 0.0042 mg L^{-1} d^{-1} of phosphorus.

4. Discussion

Treatment with FLMs decreased turbidity and SS and promoted water transparency. A similar result was observed in another study [25]. With roots anchoring into the sediment and leaves floating on the water surface, FLMs can greatly attenuate wave and current energy, resulting in increased sedimentation, reduced sediment resuspension, and decreased turbidity and SS [44]. Wave height was higher in the OA than in the area covered by FLMs. A previous study [25] found that with reduced resuspension, phosphorus fluxes from the sediment to the water column were also depressed, and thus, internal phosphorus loading was reduced.

FLMs affected both the species composition and abundance of algae. Different algal species showed different responses to the presence of FLMs. In our study, most of the planktonic algae, i.e. blue-green algae (*Merismopedia*, *Chroococcus*, and *Phormidium*), cryptomonads (*Cryptomonas* and *Chroomonas*), green algae (*Actinastrum*, *Chlorella* and *Scenedesmus*), and centric diatoms (*Cyclotella* and *Aulacoseira*) were largely distributed in the OA, while some metaphytic species (*Navicula*, *Cymbella*, and *Gomphonema*) were primarily found in areas covered by FLMs. Many laboratory studies have demonstrated an allelopathic effect of macrophytes on *Scenedesmus*, *Chlorella*, and *Microcystis* [23,45–49]. Other studies showed that epiphytic and planktonic species of cyanobacteria had differential responses to the allelopathic activity of *Stratiotes aloides* [50]. *S. aloides* extract reduced the growth of planktonic cyanobacteria but did not affect epiphytic cyanobacteria.

The FLMs profoundly improved the water transparency. This may primarily be attributed to the reduced populations of algae, as shown by chlorophyll-a concentration, algal density, and algal biomass. The values of these indices in water covered by FLMs were much lower and more stable than in the OA on average, though this difference was not consistent.

Several mechanisms could account for the inhibition of algae quantity by FLMs, e.g. nutrient competition, shading effect, and allelopathy. However, the algal density, biomass, and chla concentration in the OA within the large enclosure were mostly and on average greater than in areas covered by FLMs, although the nutrient concentrations were similar. Thus, nutrient competition may not be responsible for the low algal abundance in water covered by FLMs as observed in this study.

The light intensity in the water covered by *T. uncinata* was extremely low. Though the light intensity was comparable between site CO and site NY, the total radiation per area in the enclosure with *N. tetragona* was less than that in the OA because 50% of the surface was shaded by leaves. This finding suggests that shading may explain the low algal biomass in site T1, T2, and NY.

Other mechanisms may also account for the inhibition. An antagonistic relationship between chlorophyll-a concentrations and the growth areas of *Trapa* was presumed in a shallow eutrophic lake in Japan [21]. Similarly, it was revealed that *Egeria densa* and *Potamogeton illinoensis* were able to suppress phytoplankton growth through biological mechanisms when competition for nutrients and physical effects were excluded, providing evidence for allelopathic effects of submerged macrophytes [51]. However, we have no direct evidence for an allelopathic effect of FLMs. Because of the hypoxia and lack of food in the covered water, zooplankton might be less abundant than in the OA, according to the studies by Fontanarrosa et al. [52] and also some of our zooplankton analytical results.

Use of submerged macrophytes for habitat recovery is less practical in shallow eutrophic lakes, although it is thought to be an important strategy for the management of shallow lakes [53]. For example, we failed to recover submerged macrophytes from the large enclosure in 2009; the lack of growth could be attributed primarily to poor light conditions. Eutrophic lakes typically have low water transparency caused by high algal abundance. On the other hand, large shallow lakes are susceptible to sediment resuspension [54], which may redistribute settled materials and increase water turbidity, thus affecting light conditions [55]. Submerged macrophytes do not survive well under such conditions according to the theory of light compensation depth for submerged macrophytes [56,57]. Apart from light levels, wave action and nutrient levels also limit the recovery of submerged macrophytes [58].

Huang et al. [25] proposed that the growth of FLMs provides an alternative method for vegetation restoration. It is more practical in turbid, shallow, eutrophic lakes, especially under conditions that do not permit the survival of submerged macrophytes. Theoretically, the reduction of wave action, turbidity, and nutrient concentrations and the improved water transparency by FLMs may also provide good conditions for the restoration of submerged macrophytes. Nonetheless, the nutrient decrease and the inhibition of phytoplankton were not distinct as we expected in our study. There was no replicate in our experiments. More evidence is needed to assess the planting of FLMs as a management tool.

Notably, the DO was inversely correlated with FLM coverage because the leaves hamper gas exchange between the water and the atmosphere. FLMs also inhibit convection and lead to the depletion of DO at the lake bottom. In OA, algal photosynthesis increased DO in the water column. Recovery of FLMs may not be possible in lakes in which the water contains high concentrations of ammonia or oxygen-consuming materials.

5. Conclusions

FLMs can significantly improve water transparency by their inhibition of algal quantity. However, the decrease in nutrient concentration was not as obvious as expected in this study, both because of the limited absorption ability of the plants and because of the large fluctuation in the nutrient concentration in the lake water. The inhibitory effects on phytoplankton were mainly due to the shading effect of FLMs. FLM treatments promoted changes in the phytoplankton community composition, benefitting metaphytic species adapted to higher transparency. Planting FLMs to produce a clear water state provides an alternative management method for turbid waters in subtropical shallow eutrophic lakes where submerged macrophytes cannot survive. However, further studies are necessary to more clearly account for the benefits and drawbacks of this approach.

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