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Improvement of water quality by sediment capping and re-vegetation with *Vallisneria natans* L.: A short-term investigation using an *in situ* enclosure experiment in Lake Erhai, China

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ABSTRACT

This study evaluated the effects of sediment capping with local unpolluted soil and re-vegetation with *Vallisneria natans* on the water quality in a two-month *in situ* experiment in a mesotrophic lake. Water quality was improved substantially by sediment capping whether alone or along with re-vegetating using *V. natans*, although varied greatly with time. Sediment capping was an effective technique to reduce nitrogen (N) and phosphorus (P) release from fertile sediment, which thus inhibited phytoplankton growth in the water column. Re-vegetation using *V. natans* helped to improve water quality slightly more. Sediment capping alone or along with re-vegetating also changed interactions of the water quality parameters substantially. Compared to the control, sediment capping alone enforced the relationships between *Chl-a* and both SD and *K*, but along with re-vegetation weakened the relationships. In addition, sediment capping alone enforced the relationship between *Chl-a* and N-NH_4, but with or without along with the re-vegetation the relationships between *Chl-a* and both TN and N-NO_3 decreased.

1. Introduction

Sediment can be a sink or source for nutrients in the water column, depending on sediment fertility and other characteristics (Søndergaard et al., 2003, 2013). Release of nitrogen (N) and phosphorus (P) from sediments into the water column plays a critical role in the nutrient dynamics of shallow lakes (Beutel, 2006; Granéli, 1999; Scharf, 1999). Overloading of nutrients into lakes leads to deterioration of the ecosystem and excessive accumulation of N and P in the sediment. Excess nutrients in the sediment, in turn release into the water column and support the growth of phytoplankton, which then impedes the recovery of submerged vegetation and delays water quality improvement for decades, even when external nutrient loading has stopped (Søndergaard et al., 2003, 2013). Therefore, reducing nutrient release from sediments and re-vegetation of submerged macrophytes are proposed as a management technique for shallow eutrophic lakes (Scheffer et al., 1993; Smith and Schindler, 2009).

Many approaches including hydraulic flushing (Turner et al., 1983; Beutel, 2006), chemical injection (Wauer et al., 2005), precipitation of phosphorus by metal salts (Reitzel et al., 2006), sediment dredging (Zhong et al., 2008) and sediment capping (Gibbs and Özkundakci, 2011; Lin et al., 2011) have been employed to control N and P release from sediments in specific lakes worldwide. Among these approaches, sediment capping was considered as an effective technique that used Phoslock^{TM} (Robb et al., 2003), iron-, aluminum- and calcium-based chemicals (Cooke et al., 1993; Wauer et al., 2005) to create an inactive barrier between the sediment and the overlying water. Sediment capping not only blocks nutrient flux but also reduces resuspension of soft sediment and alters the physico-chemical properties of sediment–water interface, thus retraining nutrients in the sediment (Kim et al., 2007; Simpson et al., 2002).

However, a variety of negative effects of chemical sediment capping have been observed, such as an increase in pH value, a reduction in oxygen, and toxic effects to some aquatic

Abbreviations: N-NO_3, nitrate nitrogen; N-NH_4, ammonium nitrogen; TN, total nitrogen; DIN, total dissolved nitrogen; TP, total phosphorus; SRP, soluble reactive phosphorus; *Chl-a*, chlorophyll a; SD, Secchi-depth; *K*, light attenuation coefficient; *T*, temperature; DO, oxygen concentration; PAR, underwater photosynthetic active radiation.

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animals (Vopel et al., 2008). In pristine areas, soil particles erode from unvegetated land and are transported to lakes by rain and floods. They bury organic matter, reverse anoxic condition and retain nutrients in the sediment, an important mechanism for maintaining water quality in natural water systems. Therefore, local unpolluted soil, which is more economically and ecologically friendly, may be a potential alternative to chemicals for sediment capping (Pan et al., 2011).

Submerged macrophytes play a key role in maintaining a clear water state of eutrophic shallow lakes. They improve the water quality by ways of absorbing nutrients, stabilizing sediment, sheltering algae-filtering zooplanktons and competing with phytoplankton for light and nutrients (Scheffer et al., 1993; Qiu et al., 2001). Rooted submerged macrophytes absorb nutrients by both shoots and roots, but the relative importance of sediment *versus* overlying water as N and P sources for the plants varies among species and depends on nutrients concentrations in the water and sediment (Madsen and Cedergreen, 2002; Xie et al., 2005; Cao et al., 2011). The submerged macrophyte *Vallisneria natans* is a perennial species with a large amount of roots, that prefers mesotrophic to eutrophic water. It can absorb significant amounts of nutrients from the sediment by roots, and thus has potential for use in re-vegetation efforts in degraded lakes.

In this study, we conducted an *in situ* enclosure experiment in a littoral zone with fertile sediments, using local unpolluted soil as a sediment cap, with or without re-vegetation using of *V. natans* in order to control nutrient release from sediments and thus improve the water quality. Specifically, two hypotheses were tested: (1) sediment capping can help to improve the water quality by the inhibition of N and P release; (2) re-vegetation of *V. natans* could improve the water quality further due to its water cleaning effects.

2. Materials and methods

2.1. Experimental site

The experiment was carried out in Haichao Bay in the northern part of Lake Erhai (25°52′ N, 100°06′ E) in the subtropic Yunnan Plateau, China (Fig. 1). The lake has a mesotrophic status, a moderate water depth (max. 20.5 m, mean. 10.5 m) and a total area of 249.8 km^2. Submerged vegetation once covered about 40% of the water surface in 1980s and sharply declined to less than 10% of the water surface in 2002–2003, causing the water clarity to become turbid (Fu et al., 2013). Lake Erhai has been fertilized by agricultural runoff and rural sewage in the past years, but the external nutrient input has not continued to increase due to changes in agricultural crops that consume less fertilizer. A National High Technology Research and Development Program of China has been implemented recently in Lake Erhai, which aims to reduce external nutrient loading as well as to improve the water quality by ecosystem management. The experimental area, Haichao Bay, had thick soft fertile sediment, which has a high potential to release nutrients into the water column by wind-induced sediment resuspension. Total nitrogen (TN) and phosphorus (TP) were 2354–6174 mg kg^{-1} and 418–1108 mg kg^{-1}, respectively, in the surface sediments (about 20 cm thick). Fe/Al-P was the main form of TP and inorganic-N was the main form of TN, consisting to 43% of TN (Zhao et al., 2013a, 2013b; Ni and Wang, 2015).

2.2. Experimental design

Twelve enclosures (3 m in diameter, 4 m in height) were constructed of water proof polyvinyl chloride (PVC) textile fastened to steel tubes, creating an enclosure volume of ~17.7 m^3 and were located 100 m offshore where water depth was ~2.5 m during the experimental period (Aug. 20–Oct. 22, 2013). The lower and upper parts of the enclosures were directly open to sediment and air, respectively, for free exchange of nutrients and air. Lower edges of the surrounding PVC textile were tied with cobblestone-filled net and buried 30 cm beneath the sediment surface, so as to isolate water inside the enclosures from their surrounding water (Fig. 1). Before the beginning of treatments, the PVC textile was dropped down to 50 cm beneath the water surface for free exchange between the internal and external water of the enclosures for 5 days, and all submerged macrophytes and fishes were removed from the enclosures.

There were three treatments (control, sediment capping [Treatment 1], sediment capping + re-vegetation with *V. natans* [Treatment 2]). At the beginning of the experimental treatments, the PVC textile of each enclosure was pulled up to 100 cm above water surface. For the sediment capping, local unpolluted soil was collected from the subsurface layer (30 cm beneath earth surface) of a red soil type found on a pristine hillside near Lake Erhai. The chemical composition of red soil consists of, mass %, Al_2O_3 = 10–13%, Fe_2O_3 = 6–8%, N = 0.05–0.07%, and P = 0.015–0.019% (Zhu et al., 2007). The soil was ground into small granules (diameter < 5 mm), air-dried for 2 weeks, and filled to cover the sediments to a ~5 cm thickness. For the re-vegetation treatment, seedlings (~40 cm in height) were collected from Lake Erhai and transplanted uniformly to sediments at a density of 600 seedlings per enclosure prior to the capping procedure.

2.3. Measurements of physico-chemical parameters of water in the enclosures

Measurements of physico-chemical parameters of water in the enclosures were done on Aug. 20, 2013, one day before the beginning of the treatments, not taken due to sedimentation of the capping soil for 2 weeks, on Sept. 3 and then at one-week intervals until Oct. 22, 2013. Water samples were collected by a Patalas-Schindler sampler; each sample was a mixture of subsamples collected from three layers (surface, middle and bottom). All water samples were stored on ice immediately and transported to the laboratory for analysis of *Chl-a*, TN, total dissolved nitrogen (DIN[1]), nitrate nitrogen (N-NO_3), ammonium nitrogen (N-NH_4), TP and soluble reactive phosphorus (SRP) according to methods described by Eaton et al. (1995). For the DIN measurement, water was filtered through a 0.45 μm Whatman GF/C glass fiber filter and analyzed using the same method for TN.

Secchi-depth (SD), temperature (T), pH, dissolved oxygen concentration (DO) and underwater photosynthetic active radiation (PAR) were measured *in situ* in water in the enclosures. SD was measured by a 30-cm diameter Secchi-disk. T, DO and pH were measured by a multifunctional YSI meter (Yellow Springs Instruments, Ohio, US). PAR was measured at water depths of 0, 0.5, 1.0, 1.5, and 2.0 m using an underwater radiation sensor (UWQ-8342) connected to a data logger (Li-1400; Li-Cor Company, Lincoln, NE, U.S.A). Light extinction coefficient of the water column (K) was calculated based on the equation: $K = -\ln (I_d/I_s)$, where I_d is PAR at water depth d and I_s is PAR at the water surface (Duarte et al., 1986).

2.4. Statistical analysis

Mean values and standard errors of water quality parameters were calculated from replicates within each treatment on one sampling time to understand differences between treatments.

[1] For DIN measurement, water was filtered through a 0.45 μm Whatman GF/C glass fiber filter; the filtered water was retained for total dissolved nitrogen analysis by the same method of TN measurement.

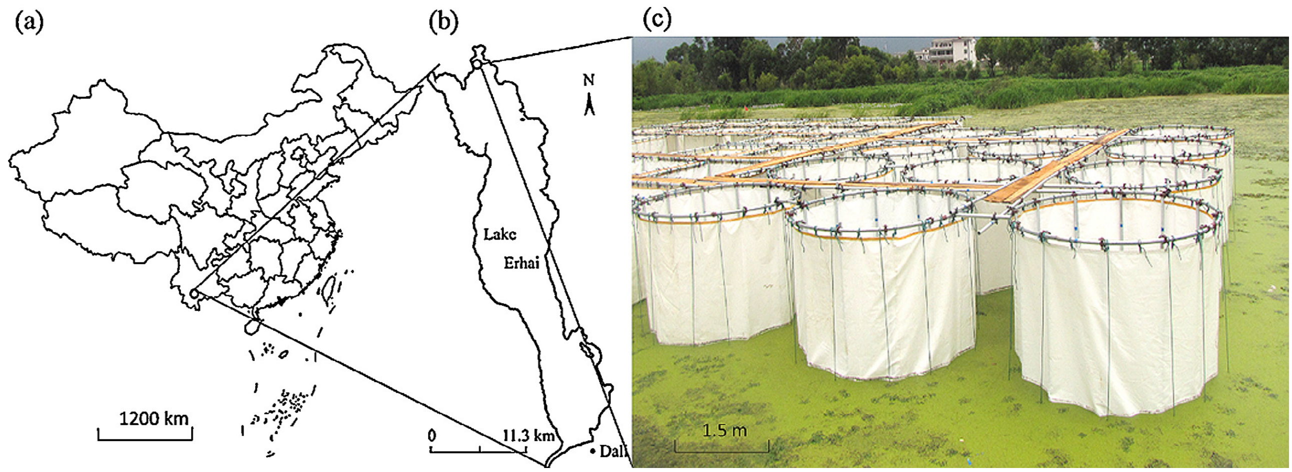

Fig. 1. Map location of the experimental site (a, b) and picture of the experimental enclosures (c).

Pearson's correlation analysis was performed for pairs of water quality parameters (TN, DIN, N-NH_4, N-NO_3, TP, SRP, SD, *K*, *Chl-a*) using SPSS 17.0 software (SPSS, Chicago, IL), with significance set at $p < 0.05$.

3. Results

3.1. Changes in water quality parameters

Water temperature was 18.5–23 °C and the pH value was 7.5–9.0 in all treatments during the experimental period (Table 1). Water quality was improved substantially by the treatments and varied greatly with time. Sediment capping alone increased SD, decreased *K*, and decreased *Chl-a*, TN, DIN, TP and SRP compared to the control. Re-vegetation of *V. natans* enforced the effects of sediment capping on SD and SRP, but diminished the effects on N-NO_3, N-NH_4 and TP (Figs. 2–4). In all three treatments, SD varied greatly and showed an unimodal response with time, increasing by 13–48% from 3-Sept. to 17-Sept., reaching the highest levels (108–148% of the values in 3-Sept.) in the subsequent one week and then decreasing rapidly, to about half of the highest values by the end of the experiment (Fig. 2A). Time-dependent changes in *K* showed a trend inverse of SD, decreasing slightly from 10-Sept. to 24-Sept. and then increasing gradually until the end of the experiment (Fig. 2B). Although *Chl-a* was closely correlated to SD (Table 2; $r = -0.599$, $p < 0.01$ in Control; $r = -0.812$, $p < 0.01$ in Treatment 1; $r = -0.299$, $p > 0.05$ in Treatment 2) and *K* (Table 2; $r = 0.750$, $p < 0.01$ in Control; $r = 0.803$, $p < 0.01$ in Treatment 1; $r = 0.317$, $p > 0.05$ in Treatment 2), *Chl-a* had different time-dependent changes (Fig. 2C). Amounts of *Chl-a* were ~8 $\mu g\,L^{-1}$ at the beginning of the experiment and then showed diverse time-dependent responses in the three treatments. In the control, *Chl-a* increased dramatically by a factor of 5 times from 20-Aug. to 3-Sept., decreased from 43 $\mu g\,L^{-1}$ to 24 $\mu g\,L^{-1}$ from 3-Sept. to 24-Sept., and then increased to 60 $\mu g\,L^{-1}$ from 24-Sept. to 8-Oct. until stabilizing at ~53–60 $\mu g\,L^{-1}$ until the end of the experiment (Fig. 2C). With sediment capping alone, *Chl-a* was low (6–14 $\mu g\,L^{-1}$) from 20-Aug. to 1-Oct. and then increased by a factor of two in one week and maintained the high levels (24–34 $\mu g\,L^{-1}$) from 8-Oct. until the end of the experiment (Fig. 2C). With sediment capping together with re-vegetation using *V. natans*, *Chl-a* increased dramatically by a factor of 8 from 20-Aug. to 3-Sept., decreased to 15 $\mu g\,L^{-1}$ in one week, maintained low levels (10–17 $\mu g\,L^{-1}$) from 10-Sept. to 15-Oct., and increased to 26 $\mu g\,L^{-1}$ in the last week of the experiment (Fig. 2C).

The concentrations of TN, DIN, N-NO_3, N-NH_4, TP and SRP changed greatly with time in the three treatments (Figs. 3 and 4). The TN concentrations increased gradually from 0.65 $mg\,L^{-1}$ to 1.56 $mg\,L^{-1}$ with time in the control, but varied between 0.53 $mg\,L^{-1}$ and 0.97 $mg\,L^{-1}$ and changed synchronically in the other two treatments (Fig. 3A). The TP concentrations were initially low (0.029 $mg\,L^{-1}$) with a tendency to increase and show a bimodal response to time, with peaks at 17-Sept. (0.102 $mg\,L^{-1}$) and 1-Oct. (0.122 $mg\,L^{-1}$) in the control, but maintained a relatively stable level between 0.03 $mg\,L^{-1}$ and 0.10 $mg\,L^{-1}$ during the entire experimental period except for a peak of 0.12 $mg\,L^{-1}$ in 1-Oct. in the other two treatments (Fig. 4A). With respect to DIN, N-NO_3, N-NH_4 and SRP, each changed synchronically in the three treatments for the majority of the experiment; N-NO_3 concentrations tended to increase, N-NH_4 concentrations tended to decrease, and DIN and SRP concentrations varied irregularly with time (Figs. 3 and 4).

3.2. Correlations among water quality parameters

Sediment capping alone or together with re-vegetation of *V. natans* changed the correlations of water quality parameters substantially. Compared to the control, the other treatments weakened the relationship between *Chl-a* with both TN and N-NO_3, but sediment capping alone enforced the relationship between *Chl-a* and N-NH_4. *Chl-a* significantly correlated with TN ($r = 0.638$, $p < 0.01$) and N-NO_3 ($r = 0.546$, $p < 0.01$) in the control treatment, TN ($r = 0.339$, $p < 0.05$), N-NO_3 ($r = 0.425$, $p < 0.05$) and N-NH_4 ($r = -0.620$, $p < 0.01$) in the sediment capping treatment, and N-NO_3 ($r = 0.338$, $p < 0.05$) in the sediment capping and re-vegetation treatment (Table 2). Correlations between *Chl-a* with both TP and SRP were insignificant. Compared to the control, the other treatments had weaker relationships between TN and TP, but stronger relationships between soluble nitrogenous compounds and SRP. Significant correlations were observed between TN and TP ($r = 0.439$, $p < 0.05$) in the control treatment, DIN and SRP ($r = -0.637$, $p < 0.01$) in the sediment capping treatment, and N-NO_3 and SRP ($r = -0.373$, $p < 0.05$) and N-NH_4 and SRP ($r = 0.352$, $p < 0.05$) in the sediment capping and re-vegetation treatment (Table 2).

Compared to the control, sediment capping alone strengthened the relationships between *Chl-a* and both SD and *K*. *Chl-a* significantly correlated with SD and *K* in the control ($r = -0.599$, $p < 0.01$; $r = 0.750$, $p < 0.01$) and the sediment capping alone treatment ($r = -0.812$, $p < 0.01$; $r = 0.803$, $p < 0.01$), but not in the sediment capping and re-vegetation treatment (Table 2). Compared to the control, sediment capping alone or together with re-vegetation had weaker relationships between SD and *K* (Table 2; $r = -0.796$, $p < 0.01$ in Control; $r = -0.701$, $p < 0.01$ in Treatment 1; $r = -0.587$, $p < 0.01$

Table 1
Water quality parameters in the enclosures with different treatments.

Water quality parameters	Control (*X*)	Capping alone (*Y*)	Capping and re-vegetation (*Z*)
T (°C)	18.8–23.0	18.8–23.0	18.8–23.0
DO (mg L^{-1})	6.0–9.4	7.0–9.4	5.9–10.4
pH	7.7–8.5	7.6–8.8	7.6–8.3

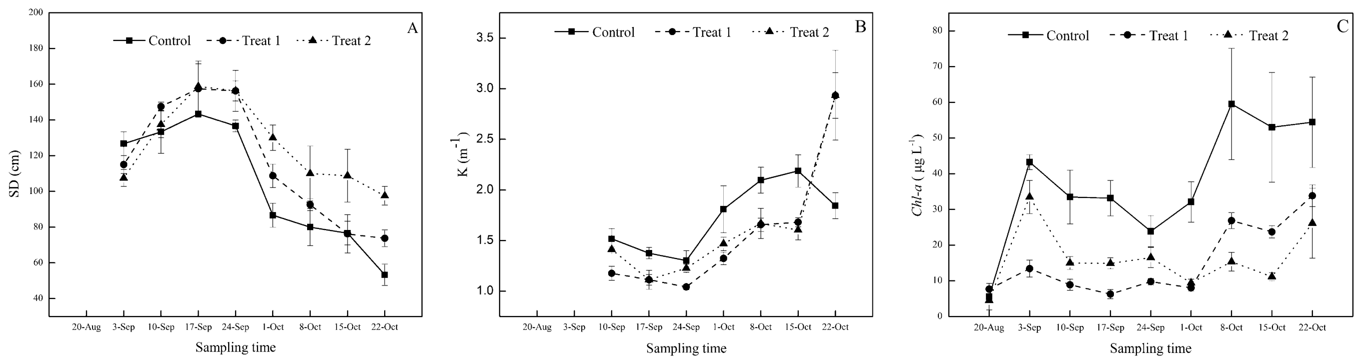

Fig. 2. Effects of the treatments on SD, *K* and *Chl-a* contents in water of the enclosures. Treat 1 indicates sediment capping. Treat 2 indicates sediment capping together with re-vegetation of *V. natans*. Values are mean ± S.E.

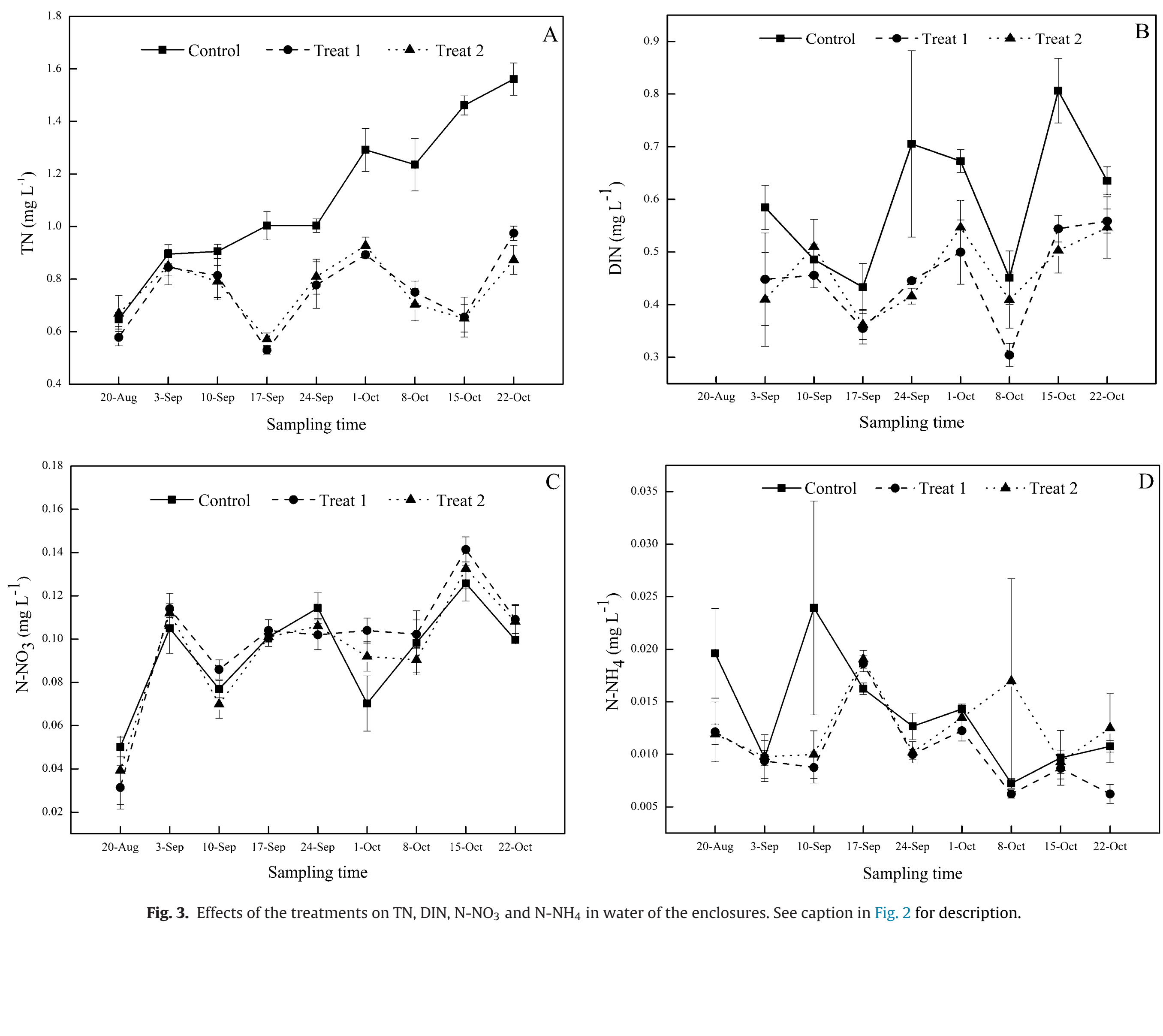

Fig. 3. Effects of the treatments on TN, DIN, N-NO_3 and N-NH_4 in water of the enclosures. See caption in Fig. 2 for description.

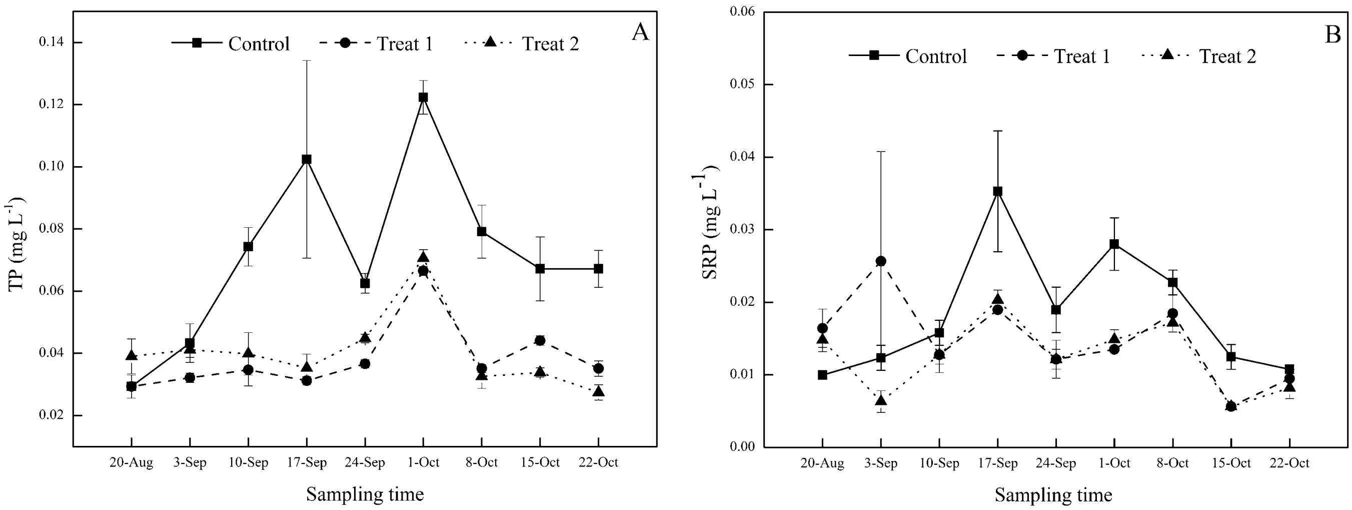

Fig. 4. Effects of the treatments on TP and SRP in water of the enclosures. See caption in Fig. 2 for description.

Table 2
Pearson's correlation coefficient matrix of water quality parameters ($n = 21$ for Control; $n = 28$ for Treat 1 and Treat 2).

		TN	DIN	$N\text{-}NO_3$	$N\text{-}NH_4$	TP	SRP	SD	*K*
Control	*Chl-a*	0.638**	0.017	0.546**	−0.375	0.204	−0.286	−0.599**	0.750**
	TN		0.356	0.404*	−0.416*	0.439*	−0.306	−0.903**	0.723**
	DIN			0.310	−0.141	−0.024	−0.255	−0.297	0.170
	$N\text{-}NO_3$				−0.433*	0.019	−0.252	−0.020	0.097
	$N\text{-}NH_4$					−0.002	0.031	0.411*	−0.381
	TP						−0.049	−0.073	0.019
	SRP							0.292	−0.257
	SD								−0.796**
Treat 1	*Chl-a*	0.339*	0.112	0.425*	−0.620**	−0.018	−0.092	−0.812**	0.803**
	TN		0.288	0.210	−0.492**	0.277	0.017	−0.243	0.455*
	DIN			0.233	−0.199	0.231	−0.637**	−0.299	0.360
	$N\text{-}NO_3$				−0.175	0.267	−0.065	−0.518**	0.203
	$N\text{-}NH_4$					−0.053	0.110	0.542**	−0.516**
	TP						−0.129	−0.214	−0.133
	SRP							0.131	−0.349
	SD								−0.701**
Treat 2	*Chl-a*	0.226	−0.158	0.338*	−0.192	0.007	−0.297	−0.299	0.317
	TN		0.297	0.115	0.001	0.160	−0.234	−0.367**	0.461**
	DIN			0.046	0.047	−0.001	−0.243	−0.154	0.331
	$N\text{-}NO_3$				0.053	−0.250	−0.373*	−0.251	0.060
	$N\text{-}NH_4$					−0.135	0.352*	−0.108	−0.044
	TP						−0.062	0.066	−0.154
	SRP							0.365*	−0.466*
	SD								−0.587**

* Significance <0.05.
** Significance <0.01.

in Treatment 2). Moreover, SD and *K* were more closely correlated with nitrogenous compounds than phosphorous compounds. SD and *K* significantly correlated with TN in the control, with $N\text{-}NH_4$ in the sediment capping treatment, and with TN and SRP in the sediment capping and re-vegetation treatment.

4. Discussion

The present study revealed that sediment capping with local unpolluted red soil alone or together with re-vegetation using *V. natans* could improve water quality substantially as indicated by increased water clarity and decreased *Chl-a* and nutrient concentrations in the water column. Decreased TN and TP in the water column suggested that there was less N and P being released from the fertile sediments, and thus supported lower *Chl-a* in the water column, as growth of phytoplankton is generally limited by N and/or P availability in freshwater systems (Søndergaard et al., 2013; Scheffer et al., 1993; Smith and Schindler, 2009). In laboratory studies, many factors could affect N and P release from sediment, *e.g.*, a high pH value and temperature, as well as resuspension and low redox potential at the sediment–water interface (Søndergaard et al., 2003; Vopel et al., 2008). The soil used for sediment capping was rich in oxidized iron and aluminum (Pan et al., 2011; Zhu et al., 2007), and dried and ground into small granules in order to increase the porosity and surface area. It not only counteracted the low redox potential but also enhanced N and P uptake from the fertile sediment, and thus locked N and P in the sediment. Al- and Fe-based salts strongly bind phosphorus and form precipitates and complexes, leading to a high P retention in the sediment and less P release from the sediment into the overlying water (Cooke et al., 1993; Reitzel et al., 2006). Some other materials, such as sand, calcite and zeolite, have also been used for sediment capping in order to inhibit P release from fertile sediments (Lin et al., 2011; Kim et al., 2007). In a 24-day laboratory experiment, Kim et al. (2007) showed that sediment capping with 40 mm and 80 mm sands decreased TP concentration in water by ~24% and 48%, respectively, but did not decrease SRP concentration substantially. Sand capping was more effective for controlling non-soluble phosphorus than soluble phosphorus possibly due to filtration effects and inactive surface of the sand grains. In general, sediment capping with various materials except for zeolite was reported to be more effective in inhibiting P release than N release from fertile sediments in microcosm experiments (Gibbs and Özkundakci, 2011; Lin et al., 2011), and thus the effects need to be evaluated through *in situ* or whole lake experiments. In the present study, re-vegetation of *V. natans* improved the water quality slightly more than sediment capping alone. It is possible that the fertile sediments diminished the uptake of N and P nutrients from water by the plants (Squires and Lesack, 2003; Rooney et al., 2003) and the experimental period was not long enough for the recruitment of algal grazers (Estlander et al., 2009; Zhao et al., 2013), although submerged macrophytes were reported to improve the water quality through multiple mechanisms, such as taking up nutrients, stabilizing sediment, sheltering algae-filtering zooplanktons and competing with phytoplankton for light and nutrients (Scheffer et al., 1993; Madsen and Cedergreen, 2002).

Compared to the control, sediment capping alone or together with re-vegetation with *V. natans* kept the experimental system more stable as indicated by the variation in *Chl-a*, TN, DIN, TP and SRP. The much higher amount of *Chl-a* in the control may have contributed to the gradual accumulation of TN and TP in the water, as massive growth of phytoplankton could pump N and P nutrients from the fertile sediment into the water (Xie et al., 2003). Moreover, the variation in *Chl-a* due to algal growth and death led to transformation and greater dynamics of nutrients in the control. The experimental enclosures were located in shallow water in the windy Lake Erhai, and thus wave-induced resuspension of soft sediment might have contributed to the release of nutrients from the sediment into the water column in the control (Hamilton and Mitchell, 1997), in contrast to the other two treatments with sediment capping, which stabilized the surface sediments (Vopel et al., 2008; Lin et al., 2011). The synchronic changes in DIN, N-NO_3, N-NH_4, SRP, SD and K suggested that abiotic factors (*i.e.*, wind-induced waves) were affecting the three treatments simultaneously, in contrast to *Chl-a*, TN and TP, which were closely related to algal growth and N and P found in the algal biomass, and thus differed significantly between the control and the other two treatments. Compared to the control, the soluble nutrients consisted to 40–85% of TN and 20–80% TP, implying that non-algal processes were dominant in the other two treatments. The experimental systems tended to become more oxidized gradually with time in all treatments as indicated by the time-dependent trends of N-NO_3 and N-NH_4, possibly due to photosynthetic oxygen generation (5.9–10.4 $mg\,L^{-1}$) of the massive phytoplankton growth in all treatments and the oxidized red soil in two treatments (Table 1).

In general, algal growth was more responsible for the amount of TN and N-NO_3, and the amounts of nutrients in the water column were more closely correlated in the control than in the other two treatments. In a field survey of many lakes worldwide, Smith (1983) showed that N-fixing cyanobacteria became dominant when the TN:TP ratio in the water column was lower than 16, suggesting that algal growth was limited by N availability. In the present study, the TN:TP ratio (15.4) was close to 16 in the control and thus suggested the algal growth could be limited by both N and P availability, but the close relationship between *Chl-a* and TN suggested that the algal growth was more likely limited by N availability (Smith, 1983; Smith and Schindler, 2009). The prolific algal growth led to a close relationship between TN and TP due to N:P stoichiometry in algal biomass (Smith, 2006; Xie et al., 2003). In the treatments of sediment capping alone or together with re-vegetation, the TN and TP were not closely correlated, but the SRP was closely correlated with DIN or N-NO_3 and N-NH_4, implying that the algal-mediated coupling between TN and TP was weakened when algal biomass was low, while non-algal processes (*i.e.*, nutrient release induced by sediment resuspension and temperature) became prevalent and affected interactions of soluble nutrients (*i.e.*, DIN, N-NO_3, N-NH_4 and SRP). Water clarity was closely correlated to *Chl-a* in the control and the sediment capping treatments, but not in the sediment capping and re-vegetation treatment, suggesting that re-vegetation could help to improve SD by ways other than inhibition of algal growth, *i.e.*, reduced resuspension of soft sediment (Donk and Bund, 2002; Estlander et al., 2009), as re-vegetation of *V. natans* helped to improve SD in the last three weeks of the experiment.

5. Conclusions

The present study revealed that sediment capping with local unpolluted soil was effective in reducing phosphorus and nitrogen release from fertile sediment and thus inhibited the growth of phytoplankton. Re-vegetation using *V. natans* helped to improve water quality only slightly more, particularly for SD and SRP, and to stabilize conditions in terms of temporal variation in *Chl-a*, TN, DIN, TP and SRP, although long-term effects of the treatments need to be evaluated. In addition, the sediment capping alone or together with re-vegetation using *V. natans* tended to decrease the effects of algal growth-related processes on the nutrient interactions while the effects of non-algal processes increased.

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