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Experiment of emergent macrophytes growing in contaminated sludge: Implication for sediment purification and lake restoration

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ABSTRACT

Three emergent macrophytes (*Zizania latifolia* (Turcz) Hand.-Mazz., *Phragmites australis* (Cav.) Trin. ex Steud. and *Typha angustifolia* Linn.) and three different sediments from Lake Dianchi of Yunnan province, China, were studied through orthogonal pot-planting experiment in order to compare the ability of the three emergent macrophytes in dealing with the contaminated sludge and to evaluate the possibility of purifying the sediment through aquatic plant rehabilitation. The results show that the number of sprouts and biomass of all the species growing in the sediment of site 3 were higher than those growing in the sediment of sites 1 and 2; the plants growing in the sediment of site 3 also exhibited the highest root activities; in each sediment, the sequence of root activity of the species was: *Z. latifolia* > *P. australi* > *T. angustifolia*; TP content in the sediments grown with different plants reduced significantly than those of control. These results indicated that these emergent plants were able to grow well in the contaminated sediment though it is black with a strong odor. *Z. latifolia* shows the highest root activity in the sediment of site 3, from which we can deduce that this plant should be the preferred pioneer species for purifying the sediment. According to their biomass and TN content and TP content, *Z. latifolia*, *P. australi* and *T. angustifolia* retained TN 16.6, 29.8, 12.8, and TP 2.2, 3.6, 3.9 g m^{-2}, respectively. Based on the climate of Dianchi valley, these plants can be harvested twice in a year. Thus, the amount of nutrient can be removed almost doubly. Therefore, the sludge can be purified through macrophytes restoration and not through sediment dredging, which was proved to be expensive and invalid in the large shallow lakes.

1. Introduction

During the past decades, with the increase of population, development of industry and agriculture, more and more effluents of industrial, agricultural and domestic wastewater were discharged into the lakes and add their nutrient pool. Meanwhile, the anthropogenic activities, such as excessive exploitation of fisheries resources and reclamation of lakeshore wetland, reduced the nutrient output from the lakes. Therefore, a mass of nitrogen, phosphorus nutrient and organic material entered lakes and were deposited in the sediment (Søndergaard et al., 2000; Jin, 2008). Especially in the area of estuaries or the bays of the large shallow lakes, the black, foul and anaerobic sludge constitutes not only a serious internal nutrient loading but also impacts plant community development and lake restoration. This kind of unstable sediment releases much more nutrient into the overlying water both in static and in dynamic state (Qin, 2007). Jeppesen et al. (2007) found that even though the external nutrient loading was controlled thoroughly, the water column phosphorus concentrations would remain high due to the sediment nutrient pool release.

Various methods have been used to reduce internal phosphorus loading in combating water eutrophication, such as sediment removal, sediment capping, sediment oxidation, phosphorus inactivation, etc. (Sand-Jensen et al., 2008). Some of these methods are proved to be effective in some small lakes. But these methods are unfeasible in the large shallow lakes. Sediment removal is expensive; especially in large lakes it is difficult to transport and deal with the excavated sediment. Moreover, sediment resuspension and redistribution may hamper the removal of the upper soft and nutrient-rich layer. Sediment resuspension also influences the effect of chemical methods because phosphorus that bonded to aluminum or iron can still return to the water column on windy days (Kronvang et al., 2005). Meanwhile, sediment removal and capping may also eliminate the propagule bank, which would affect the subsequent restoration of macrophytes. Some of these measures would also affect the benthic animal and microbial communi-

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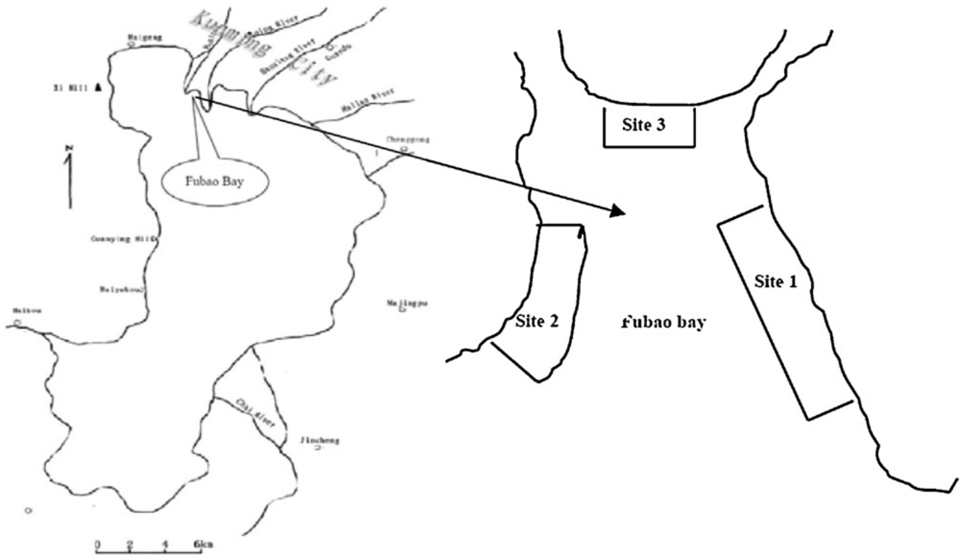

Fig. 1. Map of Lake Dianchi showing the position of the study sites.

ties, and therefore impact the self-purification abilities of the lake ecosystems (Reedyk et al., 2001).

Aquatic macrophytes, as key components in the aquatic ecosystems, are impacted by eutrophication, yet they also provide a buffer against water quality degradation and lower concentrations of nutrients both in the water column and in the sediment (Bachmann et al., 2002). Emergent macrophytes get their nutrients from sediment and therefore depress the nutrient concentration of bore water, and therefore, reduce the sediment nutrient transferring from sediment to the overlying water. Meanwhile, emergent macrophytes can also immobilize the sediment, promote deposition, restrain sediment resuspension, and therefore improve the water column transparency (Horppila and Nurminen, 2005). At the same time, macrophytes can transfer oxygen to the rhizosphere, elevate the redox potential of sediment and influence the rhizospheric microbial activities, such as nitrification, denitrification, etc. In sediment, high redox condition can also passivate the nutrients, and promote the nutrient retention in the sediment (Reina et al., 2006). Therefore, many of the emergent macrophytes have been used in nature and constructed wetlands to treat municipal, industrial, and agricultural wastewater (Kadlec and Knight, 1996; Fraser et al., 2004; Zhai et al., 2006; Asaeda and Siong, 2008), though they subjected to a number of growth constraints, such as climate, water depth, the nature of sediment, wind, waves, etc. Meanwhile, harvesting of macrophytes can not only remove substantial quantities of nutrients from the sediment but also eliminate of secondary pollution coming from the plant itself, and therefore, it has been advocated as a potential method for counteracting eutrophication (Graneli and Solander, 1988; Das and Tanaka, 2007). Loucks and Weiler (1979) calculated that harvesting half of the macrophyte biomass of Lake Wingra would reduce the spring peak of phytoplankton by 10–40%.

Therefore, ameliorating the sediment condition and eventually improving water quality through aquatic macrophyte rehabilitation and harvest was an approach that got twice the result with half the effort. We assume that even the contaminated sediment, as long as the macrophytes can grow on it, could be purified ultimately. In this paper, the reciprocities between aquatic macrophytes and the sludge were investigated through orthogonal pot experiment. The aims of this paper are:

(a) to compare the development features of the three emergent macrophytes growing in contaminative sediment and their effect on the sediment;
(b) to evaluate the possibility of rehabilitating the macrophytes to purify the sludge rather than dredging;
(c) to offer suggestions for the polluted sediment purification and lake restoration.

2. Study site

Lake Dianchi (24°40′–25°02′N, 102°36′–103°40′E), the sixth largest freshwater lake in China with an area of 300 km^2 and mean depth 4.4 m, lies in Yunnan Province, in the southwest of China (Fig. 1). The Lake Dianchi basin has a mild climate, which is characterised by short, cool dry winter, and long, warm and humid summer. With an average temperate being 15 °C in winter and 24 °C in the summer, the valley has perpetual spring-like weather which provides the ideal climate for plant. The valley has a mean annual rainfall of 1000 mm, with an annual sunshine period of 2250 h and frost-free period of 230 days.

The provincial capital city of Yunnan, Kunming, is located near the north shoreline, which is the upper reaches of the lake. With the rapid development of Kunming, industrial and municipal wastes, urban and agricultural non-point source runoff have contributed to vast amount of contaminant loadings to the lake (Table 1). Dianchi has been a hypereutrophic lake since the 1990s, and the cyanobacterial blooms annually in the most

Table 1
Some physical and chemical properties of sediment and overlying water in Lake Dianchi.

Sediment (mg g^{-1})					Overlying water (mg L^{-1})									
TN	TP	OM (%)	Eh	pH	TN	TP	NH_3-N	TDP	DP	COD	BOD	Chl-a	DO	pH
3.30	2.80	6.5	−114.5	7.37	4.45	0.317	0.69	0.053	0.033	14.59	9.69	0.149	8.38	8.81

parts of the lake. Due to the poor sunlight conditions and oxygen depletion, anaerobic sediments with unpleasant malodors have been occurred in some bays of north part, and the water quality deteriorated incessantly. Before restoration, only *Potamogeton pectinatus* Linn. remained and its coverage was not more than 0.1% in the bay. These problems have attracted the attention of both scientists and the government (Li et al., 2007).

In 2007, an aquatic vegetation rehabilitation project was implemented in Fubao bay, with an area of 1 km^2, lying in northeast of Lake Dianchi (Fig. 1). In order to cover the black and odor sediment and restore the aquatic ecosystem, the sediment near the east bank of the bay (site 1) was coming from the open water area through dredger fill, and the sediment near the west bank (site 2) was also coming from the open water area through dredger fill but the sediment was different from that of site 1. From March to June, 2007, the aquatic vegetation rehabilitation project was put in practice. In the east bank of the bay (site 1), the rehabilitated area was 35,000 m^2 (700 m × 50 m), and in the west bank (site 2) was 32,800 m^2 (410 m × 80 m). In each site, the plant rehabilitating pattern was similar: in the shallow area that near the bank restored emergent macrophytes, and whereas in the relative deep area restored submerged macrophytes, and the area of restored emergent and submerged macrophytes almost had the equal size. Through patch mosaic pattern, the emergent macrophytes restored are: *Zizania latifolia* (Turcz) Hand.-Mazz., *Phragmites australis* (Cav.) Trin. ex Steud. and *Typha angustifolia* Linn., and the submerged macrophytes restored are: *P. pectinatus*, *Ceratophyllum oryzetorum* Kom. and *Myriophyllum spicatum* Linn. As a result, the restored area of each species in east and west bank was about 5833 and 5466 m^2, respectively.

The sediment at the end of the bay (site 3) was black and stinking because of the two river of sewage discharging into the bay. In the original restoration program the serious polluted sediment was intended to dredge up, but was adjusted to in situ solidification because short of fund and without enough space to store the excavated sediment.

3. Material and methods

In preparing the pot experiment, sediment collected from the three sites (sites 1, 2 and 3) was mixed well, and was filled into pots (depth 40 cm, diameter 30 cm) to plant the three kinds of emergent macrophytes (*Z. latifolia*, *P. australis* and *T. angustifolia*), respectively. The seedlings of these macrophytes were collected from the Dianchi lakeshore wetland on condition that the seedlings of each species should be vigorous, with similar biomass and height.

The experiment was started on April 20, 2007, with each pot planting only one seedling, and each kind of sediment as well as each species was planted in 15 pots through orthogonal design (Table 2). The pots were watered every day and water level was kept at 5 cm. Three pots of each kind of sediment were not planted, as control. After 30 days growth, the sampling started with an interval of 20 days afterwards. At each sampling time, three pots of each kind of plant and each kind of sediment were collected. The number of sprouts, and above and below ground biomass of each plant in each pot were recorded. TN content and TP content (spectrophotometrically with the molybdate-blue method after digestion with H_2SO_4 and $HClO_4$) in plant samples were analyzed. AFDW (ash free dry weight, the organic fraction of the sediment, 550 °C for 4 h) and bulk density of the sediment were tested. Sediment TP content (dried, ground through a 100-mesh sieve, and through the $HClO_4$–H_2SO_4 digesting, spectrophotometrically with the molybdate-blue method) was also analyzed.

Table 2
The design of the experiment and the sediment nutrient content.

Experiment design	Site 1	Site 2	Site 3	Pot in total
Species/sediment				
Zizania latifolia	15	15	15	45
Phragmites australis	15	15	15	45
Typha angustifolia	15	15	15	45
Control	3	3	3	9
Pot in total	48	48	48	144
Sediment nutrient content				
TN ($mg\,g^{-1}$)	1.759	1.172	4.135	
TP ($mg\,g^{-1}$)	1.754	0.948	2.419	
AFDW (OM, %)	6.550	3.190	6.520	

The root is an active organ for plant nutrient absorption. Root activity directly affects the growth of the above ground parts. Measurement of root activity was according to the triphenyltetrazolium chloride (TTC) method (Li, 2000). The principle is as follows.

The TTC is a standard redox pigment, and it dissolved in water as a colorless solution. However, after being reduced, the TTC changes to triphenylformazan (TPF), which is red, stable and insoluble in water. Therefore, TTC has been widely used as a hydrogen receptor in enzyme assay. The amount of the reduced TTC can be used to express the dehydrogenase activity and as an indicator to express the root activity.

On testing, about 0.5 g root tips were placed in tube, filled with 5 ml of 0.4% TTC and 5 ml phosphate buffer (0.06 $mol\,L^{-1}$, pH 7.0). The tubes were incubated at 37 °C for about 3 h. The chemical reaction was stopped by adding 2 ml of 1 $mol\,L^{-1}$ sulfuric acid to the tube. This step was followed by extraction with TPF, which include taking the root out of the tube and placing them in the pestle, filled with 3–4 ml of ethyl acetate and a little quartz sand and then ground. The liquid phase was removed into the test tube. Ethyl acetate was added to the 10 ml level and recorded OD values with a UV–vis recording spectrophotometer at 485 nm. The OD values were used to calculate equivalent TPF concentrations with which the root activity was determined for each fresh root weight as follows:

Root vigor (TPF $mg\,g^{-1}\,FW\,h^{-1}$) = TPF reduction (mg)/root fresh weight (g)/incubated time (h)

All the statistical analyses were conducted using SPSS 12.0 for Windows. Significant values were considered at $P < 0.05$ level.

4. Results

Z. latifolia and *T. angustifolia* grew fast, and reaching their maximal height in about a month which were roughly 120 and 200 cm, respectively. But the *P. australis* grew slowly. It reached 300 cm in about 3 months. Meanwhile, the height of *Z. latifolia* and *P. australis* was not significantly different among the three sediments. The height of *T. angustifolia* growing in the sediment of site 3 was found significantly higher than those growing in the sediment of site 2 (ANOVA, $P < 0.05$).

P. australis had more sprouts than the other two species which reached more than 40 individuals at the end of the experiment, and *T. angustifolia* had fewer sprouts, with a maximum number of 9 individuals (Fig. 2a–c). In each kind of sediment, the number of sprouts of the three species had significant difference (ANOVA, $P < 0.05$). The sprout numbers of the species growing in the sediment of site 3 were higher than those growing in the sediment of sites 1 and 2 (ANOVA, $P < 0.05$). But plants growing in the sediment

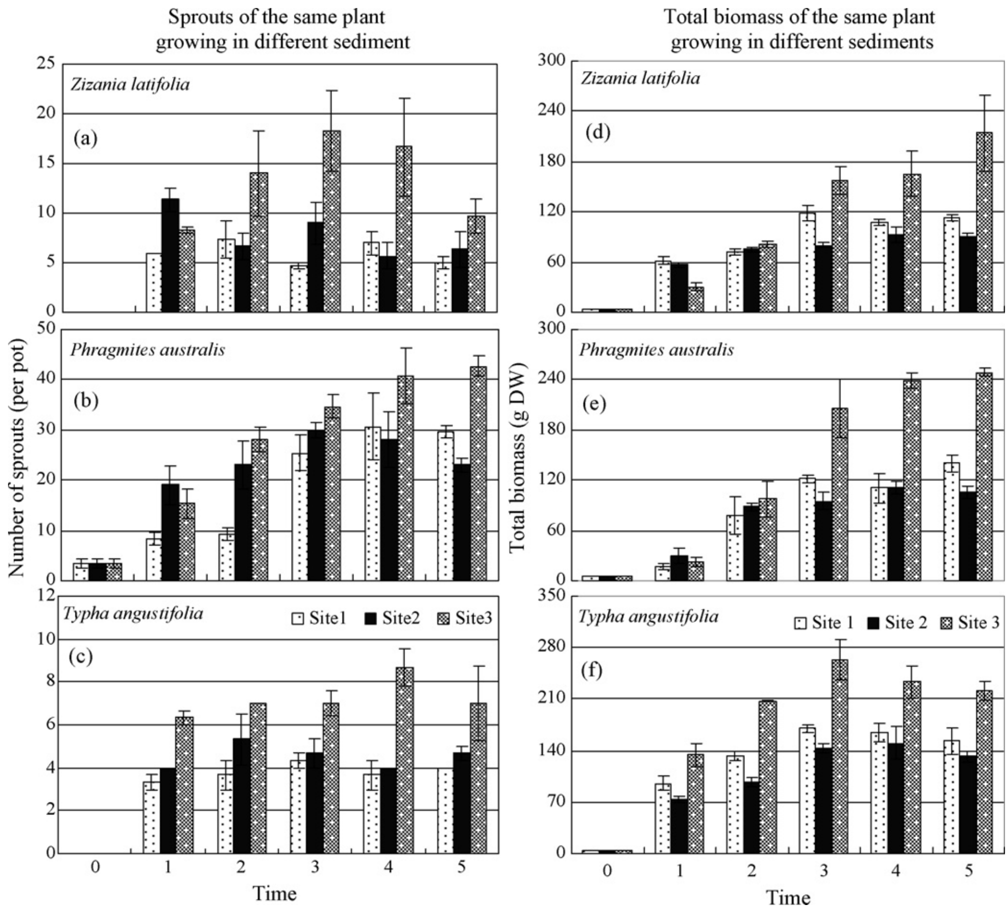

Fig. 2. Number of sprouts and biomass (per pot) of different plants (*Zizania latifolia* (a and d), *Phragmites australis* (b and e), and *Typha angustifolia* (c and f)) growing in different sediments (sites 1 (⊡), 2 (■) and 3 (▩)) (mean ± SE, $N = 3$).

of site 1 were not significantly different in sprout numbers, and the same trend was also found in the plants growing in the sediment of site 2.

The pattern of difference in biomass (root plus shoot) among different species was similar to those of sprouts (Fig. 2d–f). In the initial stage, *T. angustifolia* grew very fast, and *P. australis* grew very slowly. On average, their growth rate was 3.21 and 0.59 g DW day^{-1} in the first month. But after that stage *P. australis* growth increased rapidly, and the growth rate was 2.96 g DW day^{-1} during the 30th to the 70th day, and the growth rate of *T. angustifolia* and *Z. latifolia* was 2.28 and 1.74 g DW day^{-1} during that period, respectively. After this stage, the biomass of *T. angustifolia* declined, and the growth rate of *P. australis* and *Z. latifolia* slowed down. The maximum biomass of *T. angustifolia*, *P. australis* and *Z. latifolia* was 265, 250 and 200 g pot^{-1}, respectively.

TN content of all the plants increased in the initial month and decreased later (Fig. 3a–c). But the TN content of most of the plants increased slightly in the last stage. The TN content of all the plants growing in site 3 was higher than those growing in the sediment of sites 1 and 2 (ANOVA, $P < 0.05$), but the TN content in the plants growing in sediment sites 1 and 2 was not significantly different, respectively. The TP content in plants was all reduced during the experiment except that in site 2 their TP concentration increased in the first month and then decreased gradually (Fig. 3d–f). In *Z. latifolia*, the TP content among different sites was similar. But in *P. australis* and *T. angustifolia*, the TP content of plants growing in the sediment of site 3 was significantly higher than those growing in the sediment of sites 1 and 2 (ANOVA, $P < 0.05$).

On comparing different species growing in the same sediment, the sequence of root activity (RC) of the three species was always: *Z. latifolia* > *P. australi* > *T. angustifolia*, and the difference between

Table 3
AFDW (%) of sediment (sites 1, 2 and 3) growing with different plants (*Zizania latifolia*, *Phragmites australis*, and *Typha angustifolia*) and control (mean ± SE, $N = 3$).

	Sediment of site 1				Sediment of site 2				Sediment of site 3			
	ZL	PA	TA	Control	ZL	PA	TA	Control	ZL	PA	TA	Control
0	6.55 ± 0.30	6.55 ± 0.30	6.55 ± 0.30	6.55 ± 0.30	3.19 ± 0.09	3.19 ± 0.09	3.19 ± 0.09	3.19 ± 0.09	6.52 ± 0.09	6.52 ± 0.09	6.52 ± 0.09	6.52 ± 0.09
1	4.21 ± 0.22	4.66 ± 0.31	5.65 ± 0.32	7.83 ± 0.14	2.05 ± 0.19	2.71 ± 0.28	3.56 ± 0.32	3.55 ± 0.19	5.42 ± 0.79	3.01 ± 0.13	4.81 ± 0.19	5.83 ± 0.12
2	5.96 ± 0.34	5.72 ± 0.11	5.44 ± 0.05	8.93 ± 0.076	2.53 ± 0.24	2.91 ± 0.05	3.39 ± 0.01	4.37 ± 0.16	6.58 ± 0.09	5.42 ± 0.19	5.86 ± 0.16	7.11 ± 0.08
3	5.04 ± 0.12	6.17 ± 0.24	5.22 ± 0.12	8.95 ± 0.47	2.73 ± 0.14	2.94 ± 0.12	3.52 ± 0.14	4.64 ± 0.29	5.31 ± 0.23	6.76 ± 0.27	6.33 ± 0.16	7.31 ± 0.08
4	5.92 ± 0.37	5.43 ± 0.23	6.00 ± 0.52	9.16 ± 0.28	2.66 ± 0.12	2.69 ± 0.22	3.85 ± 0.10	4.83 ± 0.07	5.86 ± 0.11	6.59 ± 0.12	5.45 ± 0.24	7.14 ± 0.05
5	5.90 ± 0.26	6.08 ± 0.21	6.88 ± 0.28	9.77 ± 0.27	2.77 ± 0.10	2.66 ± 0.09	3.85 ± 0.14	5.21 ± 0.20	6.07 ± 0.20	7.23 ± 0.44	5.85 ± 0.10	7.76 ± 0.12

ZL: *Zizania latifolia*; PA: *Phragmites australis*; TA: *Typha angustifolia*.

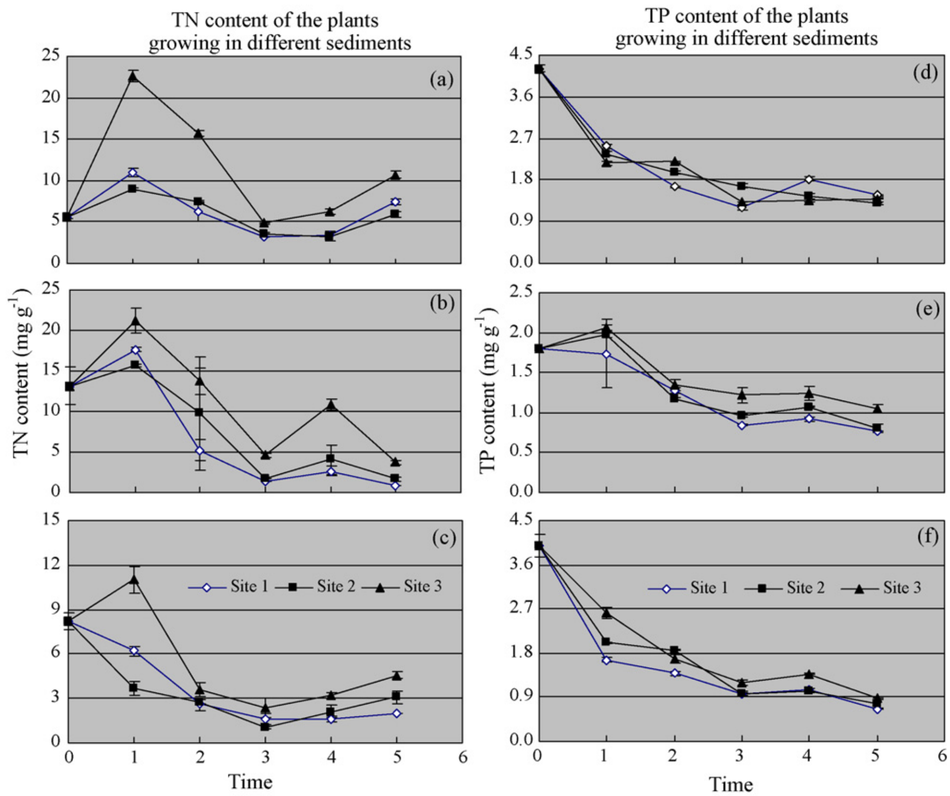

Fig. 3. TN content and TP content of different plants (shoot only) (*Zizania latifolia* (a and d), *Phragmites australis* (b and e), and *Typha angustifolia* (c and f)) growing in different sediments (sites 1 (◊), 2 (■) and 3 (▲)) (mean ± SD, $N = 3$).

species was significant (ANOVA, $P < 0.05$) (Fig. 4). On comparing the same species growing at different sediments, the plants growing in the sediment of site 3 showed the highest RC (ANOVA, $P < 0.05$). RC of *Z. latifolia* growing at the sediment of sites 1 and 2 was not found significantly different between them. RC of *P. australi* growing at the sediment of site 1 was significantly higher than those growing at the sediment of site 2. RC of *T. angustifolia* growing at the sediment of sites 1 and 2 was also not found significantly different between them.

The initial AFDW in the sediment of sites 1, 2 and 3 was 6.55, 3.19 and 6.52%, respectively. In each kind of sediment, AFDW in control was significantly higher than those growing with different plants (Table 3). In the sediment of site 1, AFDW in sediment growing with *T. angustifolia* was significantly higher than those growing with *Z. latifolia*. But in the sediment of site 3, the result was contrast with that in site1. In the sediment of site 2, the order of sediment AFDW was: the sediment growing with *T. angustifolia* > that growing with *P. australis* > that growing with *Z. latifolia*.

The initial bulk density of the sediment in sites 1, 2, and 3 was 1.09, 1.41 and 1.03 g cm^{-3}, respectively. On the whole, the bulk density in the sediment of control was not significantly different from those growing with different plants. In each kind of sediment, though growing with different plants the bulk density was almost changed slightly (Table 4).

The TP content in the sediment of control was higher than those grown with different plants (ANOVA, $P < 0.05$), and the TP in the sediment grown with plants reduced gradually though the TP concentration in the sediment of control was also reduced slightly (Fig. 5a–c). The TP content in the sediment of sites 1 and 3 grown with *T. angustifolia* was all lower than those growing with *P. australis*.

5. Discussion

P. australis had more sprouts, with more than 40 sprouts in one pot, which amounted to more than 560 m^{-2} individual. *T. angustifolia* had fewer sprouts in one pot which only amounted to 126 m^{-2} individuals. But all the species growing in the sediment of site 3 had the most sprouts. Meanwhile, comparing the total biomass of the same plant growing in different sediments, sediment of site

Table 4
Bulk density (g cm^{-3}) of sediment (sites 1, 2 and 3) growing with different plants (*Zizania latifolia*, *Phragmites australis*, and *Typha angustifolia*) and control (mean ± SE, $N = 3$).

	Sediment of Site 1				Sediment of Site 2				Sediment of Site 3			
	ZL	PA	TA	Control	ZL	PA	TA	Control	ZL	PA	TA	Control
0	1.09 ± 0.06	1.09 ± 0.06	1.09 ± 0.06	1.09 ± 0.06	1.41 ± 0.03	1.41 ± 0.03	1.41 ± 0.03	1.41 ± 0.03	1.03 ± 0.03	1.03 ± 0.03	1.03 ± 0.03	1.03 ± 0.03
1	1.03 ± 0.07	0.97 ± 0.01	1.11 ± 0.01	1.07 ± 0.03	1.45 ± 0.07	1.32 ± 0.01	1.41 ± 0.02	1.46 ± 0.01	0.97 ± 0.02	0.95 ± 0.02	1.12 ± 0.04	0.98 ± 0.04
2	1.05 ± 0.03	1.03 ± 0.05	1.02 ± 0.01	1.07 ± 0.03	1.39 ± 0.04	1.39 ± 0.01	1.30 ± 0.05	1.38 ± 0.02	0.94 ± 0.01	0.95 ± 0.04	1.11 ± 0.03	1.02 ± 0.02
3	1.00 ± 0.03	1.00 ± 0.02	1.13 ± 0.02	1.08 ± 0.01	1.33 ± 0.02	1.30 ± 0.01	1.36 ± 0.01	1.44 ± 0.02	1.09 ± 0.03	0.94 ± 0.15	1.06 ± 0.04	0.98 ± 0.01
4	1.02 ± 0.06	0.95 ± 0.01	1.11 ± 0.06	1.10 ± 0.10	1.37 ± 0.14	1.39 ± 0.02	1.23 ± 0.01	1.42 ± 0.01	0.97 ± 0.03	0.99 ± 0.00	1.12 ± 0.01	1.01 ± 0.03
5	1.03 ± 0.05	0.99 ± 0.04	1.01 ± 0.02	1.05 ± 0.04	1.30 ± 0.06	1.34 ± 0.03	1.25 ± 0.02	1.37 ± 0.00	1.15 ± 0.05	1.10 ± 0.01	1.06 ± 0.02	0.99 ± 0.08

ZL: *Zizania latifolia*; PA: *Phragmites australis*; TA: *Typha angustifolia*.

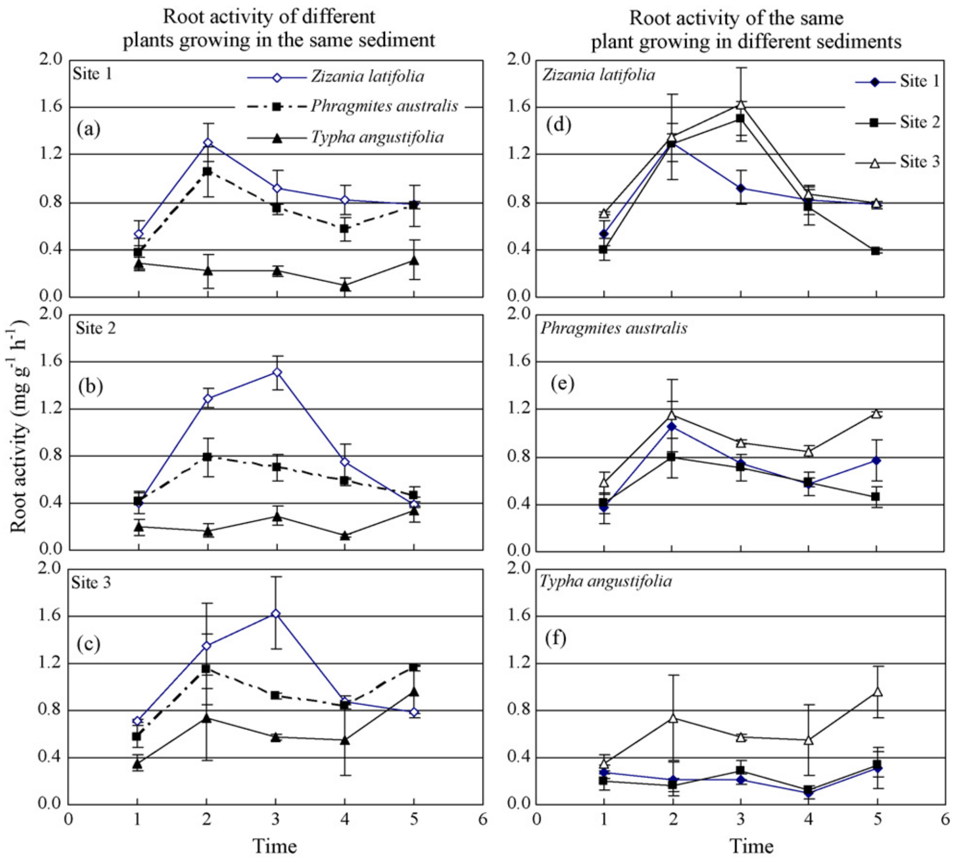

Fig. 4. Root activity of different plants (*Zizania latifolia* (◇), *Phragmites australis* (■), and *Typha angustifolia* (▲)) growing in the same sediment (sites 1 (a), 2 (b) and 3 (c)) and the same plant (*Zizania latifolia* (d), *Phragmites australis* (e), and *Typha angustifolia* (f)) growing in different sediments (sites 1 (♦), 2 (■), and 3 (△)) (mean ± SE, N = 3).

3 also sustained the highest biomass. We also found that all the plants growing in the sediment of site 3 also exhibit the highest root activities than those growing in the sediment of sites 1 and 2. From these results we can infer that the sludge, though looking black and stinking smell, can still afford the emergent plant growth that make it possible for the sediment to be purified through macrophytes rehabilitation.

Many of these emergent macrophytes are used in the nature or constructed wetlands to treat the municipal, industrial, agricultural wastewater. The treatment systems were provided to be simple, low-cost and effective in removing these pollutants (Knight et al., 2000; Cerezo et al., 2001; Zhai et al., 2006). The sewer has fewer deleterious metals, and this sludge can be purified through aquatic macrophyte rehabilitation. Aquatic macrophytes can not only assimilate the sediment nutrients directly, but they can also influence sediment interstitial pore water chemistry, particularly on the distribution of soluble reactive phosphorus. They can also create an oxygenated rhizosphere by the translocation of oxygen to the roots and increase sediment oxidation–reduction potentials. Therefore, concentrations of these redox-sensitive nutrients were reduced in interstitial waters and restrain the nutrient transfer from sediment to the overlying water (Moore et al., 1994). Therefore, the sediment can be purified through macrophytes restoration and not through sediment dredging, which was proved to be expensive and invalid in large shallow lakes (Pu et al., 2000; Kronvang et al., 2005).

The roots of *Z. latifolia* have shown the highest root activity in the sediment of site 3, by which we can deduce that this plant was able to grow well in the silt, and would be the preferred pioneer species for purifying the sediment (Chong et al., 2003; Wang et al., 2007). Roots of *P. australis* and *T. angustifolia* show relative lower root activities, therefore, the order of these species adapting this kind of sediment would be *Z. latifolia* > *P. australis* > *T. angustifolia*.

The ash free dry weight (AFDW) in all the sediments increased gradually. As plants grow, died roots and other litter mould add the sediment organic material content and the AFDW raised. But the AFDW in the sediment of control was also raised. We think that as the sediment was dug and stirred for experiment, these actions would add air to the mud which might be in favor of the microorganism growth and breeding, and therefore, increase the sediment AFDW content. The dry bulk density is related, directly or indirectly, to the softness (or hardness) of the sediment, its vulnerability to erosion, resuspension and other mechanical properties (Avnimelech et al., 2001). So we want to explore if the macrophytes would solidify the sediment and therefore decrease sediment resuspension. The sediment bulk density was in inverse proportion to its organic material content (Menounos, 1997), and if the sediment were under undisturbed, its bulk densities would increase with time (Lick and McNeil, 2001). In our experiment, though plant macrophytes would increase the sediment organic material content and therefore reduce sediment bulk density, as time passed, the sediment would solidify itself in undisturbed condition. So, eventually the bulk density changed smoothly and reduced slightly.

The biomass determines the efficiency of plants in purifying the polluted sediment, and the efficiency was also involved in the nutrient content of the plants. Shoots of *Z. latifolia* had a relative higher TN content and TP content. So based on treating efficiency, *Z. latifolia* and *T. angustifolia* would be preferred species in purifying this kind of sediment. *Z. latifolia*, *P. australis* and *T. angustifolia*

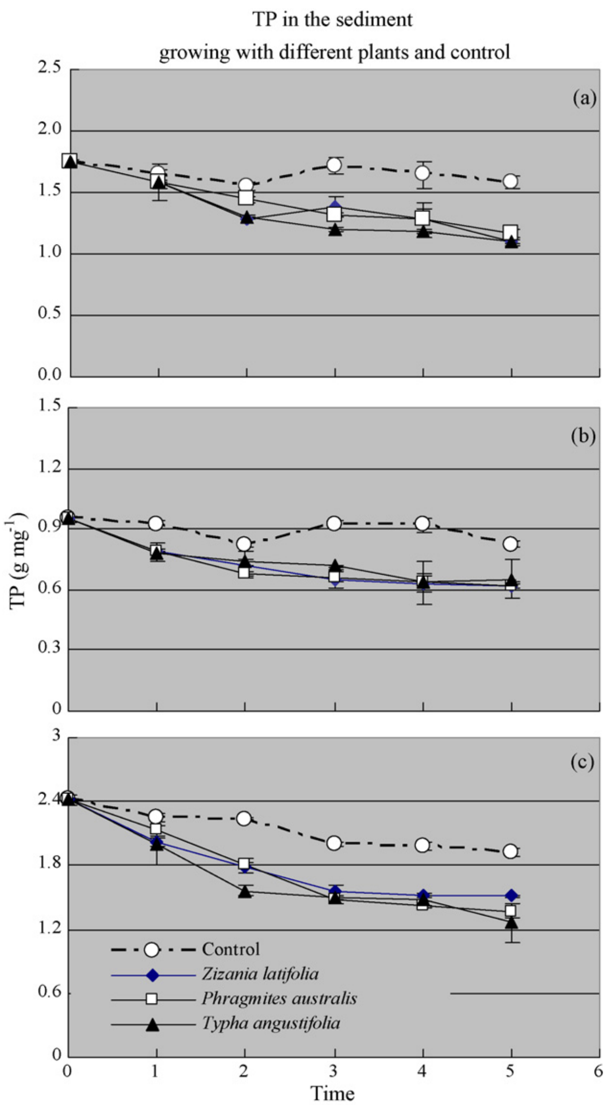

Fig. 5. TP content (mg g^{-1}) in sediment (sites 1 (a), 2 (b) and 3 (c)) growing with different plants (*Zizania latifolia* (♦), *Phragmites australis* (□), and *Typha angustifolia* (▲)) and control (○) (mean ± SD, *N* = 3).

are being used widely in different wetlands to deal with different wastewater, and were found effectively in treating many kinds of discharges (Cerezo et al., 2001; Picard et al., 2005; Huett et al., 2005; Klomjek and Nitisoravut, 2005; Das and Tanaka, 2007). Through nutrient translocation and redistribution, Asaeda and Siong (2008) found *Z. latifolia* was more efficient than other emergent plants in removing nutrients from sediment. In a subsurface flow constructed wetlands, Wang et al. (2008) found *Z. latifolia* stimulated the N_2O emission and favored to N remove from sediment because *Z. latifolia* transported more oxygen to ammonia-oxidizing bacteria.

The shoot biomass of *Z. latifolia*, *P. australis* and *T. angustifolia* growing in the pot can amount to 1549.8, 2755.2 and 2787.4 g m^{-2} DW, respectively. If they were harvested at highest biomass, according to their TN content and TP content they would remove TN 16.6, 29.8, 12.8, and TP 2.2, 3.6, 3.9 g m^{-2}, respectively. Mandi et al. (1996) thought that in a reed bed the aerial biomass exported 5–6% of nitrogen and 10–12% of phosphorus with regard to their load in the influent. Similar to our result, Ennabili et al. (1998) reported the highest total biomass of *P. australis* and *T. angustifolia* was 52.7 and 56.5 t dry weight ha^{-1} year^{-1}, respectively. They also retained in their tissues nitrogen and phosphorus at the rates of 561, 922 kg N ha^{-1}, and 72.1, 114 kg P ha^{-1}, respectively. Tanaka (2007) showed that harvesting *T. angustifolia* shoots at July and autumn removed 36.0 g N m^{-2}. Jiang et al. (2007) considered seasonal harvest of helophyte vegetation was an effective method to remove N and P from wetlands. They suggested that harvesting twice in a year would improve nutrient removal efficiency and decrease it being translocation from shoots to rhizomes. According to the climate in Kunming, these plants can be harvested twice in a year, thus the amount of nutrient can be removed almost doubly. As for the residue recycle, some of the harvest plant, such as *Z. latifolia*, was used to breed grass carp. Some of the plant was broken into pieces and mixed with manure to produce biogas for cooking, and the residue was used as organic fertilizer. Some of the harvest plants, such as *P. australis*, were used as scaffolding material to cultivate hotbed chives.

Consequently, using aquatic plants to deal with the polluted sediment and remove the nutrients through harvesting can not only improve the nature of sediment but also don't produce any harmful side effects, and the plants also play a constructive role in building a health ecosystem. Therefore, it has potential for "green" processing technology. Thus, with the research development and the technology improvement, using aquatic plants to deal with polluted sediments and to engage in lake ecological restoration will be more and more widely used. As for the species option, we suggest that the plants should have powerful roots, be fast-growing and strong adaptable indigenous species; furthermore it would be even better if the plants had some economic and scenic value as well.

6. Conclusions

Our experiments showed that: (a) the contaminated sediment, though looking black and having a strong odor, can still afford the macrophyte growth that makes it possible for the sediment to be purified through macrophyte rehabilitation. Considering biomass and TN content and TP content, *Z. latifolia* and *T. angustifolia* would be preferred species in purifying this kind of sediment; (b) from the root activity of *Z. latifolia* we can deduce that this plant was able to grow well in the sediment. The order of these species adapting to this kind of sediment would be *Z. latifolia* > *P. australis* > *T. angustifolia*; (c) contrasting with the control, the TP content in sediments grown with macrophytes reduced significantly. According to the climate in Kunming, these plants can be harvested twice a year; thus, the amount of nutrient can be removed almost doubly. Therefore, the sludge can be purified through macrophyte rehabilitation and not through sediment dredging, which was proved to be expensive and invalid in large shallow lake.

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References

Asaeda, T., Siong, K., 2008. Dynamics of growth, carbon and nutrient translocation in *Zizania latifolia*. Ecol. Eng. 32 (2), 156–165.

Avnimelech, Y., Ritvo, G., Meijer, E.L., Kochba, M., 2001. Water content, organic carbon and dry bulk density in flooded sediments. Aquacult. Eng. 25, 25–33.

Bachmann, R.W., Horsburgh, C.A., Hoyer, M.V., Mataraza Jr., L.K., Canfield, D.E., 2002. Relations between trophic state indicators and plant biomass in Florida lakes. Hydrobiologia 470, 219–234.

Cerezo, G.R., Suarez, M.L., Vidal-Abarca, M.R., 2001. The performance of a multi-stage system of constructed wetlands for urban wastewater treatment in a semiarid region of SE Spain. Ecol. Eng. 16, 501–517.
Chong, Y., Hu, H., Qian, Y., 2003. Advances in utilization of macrophytes in water pollution control. Tech. Equip. Environ. Pollut. Control 4 (2), 36–40.
Das, S.C., Tanaka, N., 2007. Estimating nitrogen budgets of Typha angustifolia by considering the regrowth shoot productivity and nitrogen content after harvesting aerial organs in different growing seasons. Landscape Ecol. Eng. 3 (2), 99–108.
Ennabili, A., Ater, M., Radoux, M., 1998. Biomass production and NPK retention in macrophytes from wetlands of the Tingitan Peninsula. Aqua. Bot. 62 (1), 45–56.
Fraser, L.H., Carty, S.M., Steer, D., 2004. A test of four plant species to reduce total nitrogen and total phosphorus from soil leachate in subsurface wetland microcosms. Bioresource Technol. 94, 185–192.
Graneli, W., Solander, D., 1988. Influence of aquatic macrophytes on phosphorus cycling in lakes. Hydrobiologia 170, 245–266.
Horppila, J., Nurminen, L., 2005. Effects of different macrophyte growth forms on sediment and P resuspension in a shallow lake. Hydrobiologia 545, 167–175.
Huett, D.O., Morris, S.G., Smith, G., Hunt, N., 2005. Nitrogen and phosphorus removal from plant nursery runoff in vegetated and unvegetated subsurface flow wetlands. Water Res. 39, 3259–3272.
Jeppesen, E., Søndergaard, M., Meerhoff, M., Lauridsen, T.L., Jensen, J.P., 2007. Shallow lake restoration by nutrient loading reduction—some recent findings and challenges ahead. Hydrobiologia 584, 239–252.
Jiang, C., Fan, X., Cui, G., Zhang, Y., 2007. Removal of agricultural non-point source pollutants by ditch wetlands: implications for lake eutrophication control. Hydrobiologia 581, 319–327.
Jin, X., 2008. The key scientific problems in lake eutrophication studies. Acta Scientiae Circumstantiae 28 (1), 21–23.
Kadlec, R.H., Knight, R.L., 1996. Treatment Wetlands. Lewis Publishers, Boca Raton.
Klomjek, P., Nitisoravut, S., 2005. Constructed treatment wetland: a study of eight plant species under saline conditions. Chemosphere 58, 585–593.
Knight, R.L., Victor, W.E., Payne Jr., Robert, E.B., Ronald, A.C., John, H.P., 2000. Constructed wetlands for livestock wastewater management. Ecol. Eng. 15, 41–55.
Kronvang, B., Jeppesen, E., Conley, D.J., Søndergaard, M., Larsen, S.E., Ovesen, N.B., Carstensen, J., 2005. Nutrient pressures and ecological responses to nutrient loading reductions in Danish streams, lakes and coastal waters. J. Hydrol. 304, 274–288.
Li, H., 2000. Principles and Techniques of Plant Physiological Experiment. Higher Education Press, Beijing, pp. 119–120, in Chinese.
Li, Q., Zhang, W., Wang, X., Zhou, Y., Yang, H., Ji, G., 2007. Phosphorus in interstitial water induced by redox potential in sediment of Dianchi Lake, China. Pedosphere 17 (6), 739–746.
Lick, W., McNeil, J., 2001. Effects of sediment bulk properties on erosion rates. Sci. Total Environ. 266, 41–48.
Loucks, O.L., Weiler, P.R., 1979. The effects of harvest removal of phosphorus on remineralized P sources in a shallow lake. In: Breck, J.E., Prentki, R.T., Loucks, O.L. (Eds.), Aquatic Plants, Lake Management, and Ecosystem Consequences of Lake Harvesting. Center for Biotic Systems, Inst. Environmental Studies, Univ. of Wisconsin, Madison, pp. 191–207.
Mandi, L., Houhoum, B., Asmama, S., Schwartzbrod, J., 1996. Wastewater treatment by reed beds: an experimental approach. Water Res. 30 (9), 2009–2016.
Menounos, B., 1997. The water content of lake sediments and its relationship to other physical parameters: an alpine case study. The Holocene 7, 207–212.
Moore, B.C., Lafer, J.E., Funk, W.H., 1994. Influence of aquatic macrophytes on phosphorus and sediment porewater chemistry in a freshwater wetland. Aquat. Bot. 49 (2–3), 137–148.
Picard, C.R., Fraser, L.H., Steer, D., 2005. The interacting effects of temperature and plant community type on nutrient removal in wetland microcosms. Bioresource Technol. 96, 1039–1047.
Pu, P., Wang, G., Hu, C., Hu, W., Fan, C., 2000. Can we control lake eutrophication by dredging? J. Lake Sci. 1 (12), 269–279.
Qin, B.Q., 2007. Principles and approach for lake ecological restoration. Acta Ecologica Sinica 27 (11), 4848–4858.
Reedyk, S., Prepas, E.E., Chambers, P.A., 2001. Effects of single Ca(OH)2 doses on phosphorus concentration and macrophyte biomass of two boreal eutrophic lakes over 2 years. Freshwater Biol. 46, 1075–1087.
Reina, M., Espinar, J.L., Serrano, L., 2006. Sediment phosphate composition in relation to emergent macrophytes in the Donana Marshes (SW Spain). Water Res. 40, 1185–1190.
Sand-Jensen, K., Pedersen, N.L., Thorsgaard, I., Moeslund, B., Borum, J., Brodersen, K.P., 2008. 100 years of vegetation decline and recovery in Lake Fure, Denmark (1). J. Ecol. 96, 260–271.
Søndergaard, M., Jeppesen, E., Jensen, J.P., Lauridsen, T., 2000. Lake restoration in Denmark. Lakes Reservoirs: Res. Manage. 5, 151–159.
Wang, C., Yang, H., Liu, G., He, S., Zhou, S., 2007. Comparative studies on purification efficiency of Zizania latifolia and Acorusca lamus in eutrophic water. J. Plant Resour. Environ. 16 (1), 40–44.
Wang, Y., Inamori, R., Kong, H., Xu, K., Inamori, Y., Kondo, T., Zhang, J., 2008. Nitrous oxide emission from polyculture constructed wetlands: effect of plant species. Environ. Pollut. 152 (2), 351–360.
Zhai, H., Liu, J., Wang, S., Wang, Y., 2006. The application of aquatic vascular macrophytes on ecological restoration of eutrophication lake. Rev. China Agricult. Sci. Technol. 8 (3), 47–52.