Ecological Engineering 83 (2015) 281–290

Contents lists available at ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

Study of the application of the water-lifting aerators to improve the water quality of a stratified, eutrophicated reservoir

Wei-Xing Ma, Ting-Lin Huang*, Xuan Li

School of Environmental and Municipal Engineering, Xi'an University of Architecture and Technology, Yanta Road 13, Xi'an, Shaanxi 710055, PR China

ARTICLE INFO

Article history:
Received 15 January 2015
Received in revised form 3 June 2015
Accepted 20 June 2015
Available online 10 July 2015

Keywords:
Cyanobacterial blooms
Reservoir
Stratification
Water-lifting aerator
Internal pollution

ABSTRACT

Cyanobacterial blooms accompanied by a release of nutrients from the bottom sediments in the stratified summer period continue to be a serious nuisance to water quality managers. Water-lifting and aeration technology can be used to inhibit algal growth while controlling internal pollutants. In this study, two water-lifting aerators (WLAs) were installed in a stratified, eutrophicated reservoir to investigate its ability to improve the water quality. The results showed that the lower water layer was directly oxygenated by WLAs. After the WLAs operated for a month, the concentration of dissolved oxygen at the bottom increased from 0 mg/L to 5 mg/L; the release of internal pollutants had effectively been suppressed. The advection generated by the circulated flow from WLAs can transport algae from the surface layer to the bottom layer. The mixing function of WLAs can control algal growth, and its mixed conduction speed gradient can effectively resist cyanobacteria floatation and reduce the competitive advantage of harmful algae, which changes the quantity and structure of the phytoplankton community. The algal cell density decreased to less than 10 million/L, with cyanobacteria accounting for only 16% of the population in the area of 10 m away from the WLA. The quantity of algae reached to 100 million/L and included Microcystis, which accounted for 91% of the population in the area of 100 m away from the WLA after three weeks of operation. The results can provide direct technical support for reservoir restoration.

1. Introduction

Thermal stratification plays an important role in the dynamics of water quality and the ecological characteristics of lakes and reservoirs (Elçi, 2008; Han et al., 2000; Rangel-Peraza et al., 2012; Wang et al., 2014). The hypolimnion becomes anoxic during the stratified period, which increases the release of pollutants from the bottom sediments and leads to the periodic deterioration of water quality (Beutel et al., 2008; Jankowski et al., 2006). Some studies of the water quality of subtropical and tropical reservoirs have found that thermal stratification can lead to anoxic conditions in the hypolimnion throughout most of the year (Hawley et al., 2006; Wang et al., 2012). Longer periods of anoxia result in the more significant deterioration of water quality as reduced species continue to diffuse out of the sediment and enter the hypolimnion. Two major countermeasures are available for the anoxic condition in the lower water layer due to stratification: one strategy overcomes water stratification using artificial mixing techniques. The other aims to directly oxygenate the water in the deep layer with hypolimnetic oxygenation systems (Antenucci et al., 2005; Bryant et al., 2011; Chowdhury et al., 2014; Soltero et al., 1994). Even though de-stratification can improve the dissolved oxygen content in the hypolimnion by mixing it oxygen-enriched surface water, hypolimnetic oxygenation is more widely applied to mitigate anoxia in the hypolimnion. This strategy retains natural stratification in order to preserve the habitat for the fauna living in the cold waters at the bottom, avoid the pollutant release that ensues from mixing the bottom sediment mixed with the surface water and reduce the operation cost (Beutel, 2006; Bryant et al., 2011; Gantzer et al., 2009; Toffolon et al., 2012).

Reservoirs have created an artificial environment conducive to algal growth, with calm waters, low light attenuation and a relatively long residence time. Toxic cyanobacterial blooms in reservoirs have become a worldwide problem (Costa et al., 2006; O'Neil et al., 2012Tarczyska et al., 2001; Te and Gin, 2011). Although any resources that are essential for algae growth can be potentially limited, available nutrient (in particular C, N, P, and Si for diatoms) and light primarily drive the change of the phytoplankton communities (Chen et al., 2009; Paerl et al., 2011a, 2011b, 2011c). Summer stratification generally favors potentially toxic cyanobacterial biomass due to their higher affinity for nutrients compared to other phytoplankton species and their ability to

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* Corresponding author. Fax: +86 029 82202729.
E-mail address: huangtinglin@xauat.edu.cn (T.-L. Huang).

http://dx.doi.org/10.1016/j.ecoleng.2015.06.022
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adjust their buoyancy and therefore exploit nutrients, light and inorganic carbon resources over a wider range (Liu et al., 2012; Wynne et al., 2008). A number of ecological engineering techniques have been used to control cyanobacteria blooms, such as altering the hydrology to enhance vertical mixing and flushing to counter the formation of cyanobacteria surface blooms (Li et al., 2013; Lundgren et al., 2013; Thackeray et al., 2006), the operation of a bypass to decrease nutrient loads and the transportation of phytoplankton from upstream areas to the reservoir (Yajima et al., 2013), and bio-manipulation by means of introducing fish to increase the grazing pressure on cyanobacteria. Unfortunately, application of these techniques in other reservoirs has several limitations; meanwhile, the reservoir management department also wants to use more effective technique to control cyanobacterial bloom in situ (Upadhyay et al., 2013).

Cyanobacterial blooms accompanied by the release of nutrients from the bottom sediments during the stratified summer period have become a common problem for most eutrophic, stratified reservoirs; these blooms seriously threaten drinking water supplies and ecological sustainability. Although the ability of artificial mixing techniques to solve the above problems has been widely confirmed, few comprehensive studies of the effects of the practical application and operating conditions of technologies to inhibit cyanobacterial blooms and the release of internal pollutants have been published, especially in China (Kim et al., 2007; Yum et al., 2008). In this research, two water-lifting aerators (WLAs) made of fiber reinforced polypropylene (FRPP) were installed in a stratified eutrophicated reservoir to improve the water quality. The WLA is a new type of artificial mixing equipment that has two major functions: mixing the lower and the upper water layers and directly oxygenating the lower water layer (Cong et al., 2010, 2011, 2009). We aimed to investigate the mixing and oxygenation capacity of this technique for controlling the algae bloom and the release of internal pollutants as well as the broader impacts on reservoir water quality.

2. Material and methods

2.1. Study site

The Shibianyu Reservoir (SBYR) is a medium-sized canyon-shaped reservoir located in a warm temperate zone approximately 35 km southwest of Xi'an city in Shaanxi province, northwest of China (Fig. 1). It was built in 1975, began to supply water to Xi'an starting in August 1990, and was used for flood control, agricultural irrigation, and power generation. The total capacity of the SBYR is 28.1 million m^3, and it supplies 30 million m^3 water to Xi'an every year. Generally, the high water level is 731 m above sea level (a.s.l.), and the low water level is 675 m a.s.l. As the city backup water resource, the water quality of the SBYR plays an important role in ensuring urban water security.

Originating from the Qinling Mountain, Shibianyu River is the main stream to supply the SBYR in Shaanxi Province (Fig. 1). This river is 30 km long with a catchment area of 132 km^2; the annual

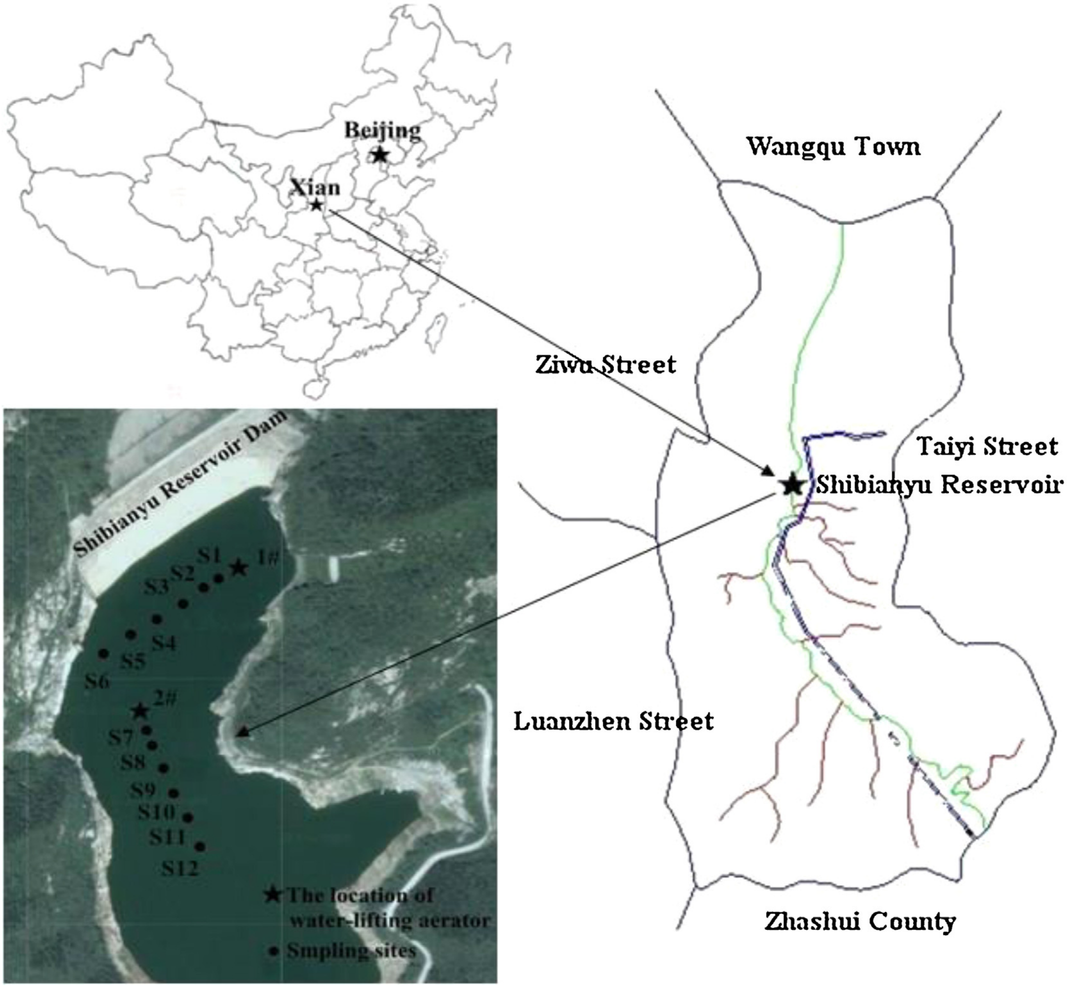

Fig. 1. Illustration of Shibianyu reservoir showing the location of the water-lifting aerators and the monitoring sites.

rainfall is 898 mm, and the runoff volume is 0.97 million m^3. The upstream and surrounding landscapes of the reservoir are largely unmodified and consist primarily of hills covered with forest that experience little human activity, which ensures that the inflow water is in good condition most of the time.

The SBYR is a stratified, eutrophicated reservoir with an annual average total nitrogen (TN) and total phosphorus (TP) concentrations of 2.8 mg/L and 0.041 mg/L, respectively (Huang et al., 2014a). Summer stratification leads to the release of internal pollutants accompanied by cyanobacteria blooms that seriously affect the reservoir water quality. Two WLAs were installed in the SBYR for this experimental study to improve the water quality. 1# WLA was located close to the dam; the water depth of this region is relatively shallow but experiences serious cyanobacterial blooms. 2# WLA was located at the deepest point of reservoir, which experiences stable summer stratification. To investigate the effect of the WLAs on the water quality of SBYR, 12 monitoring sites were set up around the WLAs. S1 to S6 were arranged along the west radial direction at distances of 10 m, 20 m, 40 m, 60 m, 80 m and 100 m from 1# WLA. S7 to S12 were arranged along the south radial direction 10 m, 20 m, 40 m, 60 m, 80 m and 100 m away from 2# WLA (Fig. 1).

2.2. Field observations

Extensive surveys of the SBYR were conducted at S7 and S12 every week starting in February 2011 to investigate the dynamic changes of the thermal regime and water quality prior to the operation of WLAs. During the operation period, the sampling frequency increased to almost once per day at all monitoring sites to monitor the effects of the WLAs on the water quality. The water depth, temperature, and dissolved oxygen content were measured at 1-m intervals from the bottom to the surface at each sampling site using a Hydro-lab DS5 (HACH Co., America) multi-parameter water quality meter. Samples were collected from each site at 5-m intervals from the surface (0.5 m under the water surface) to the bottom (0.5 m above the sediment) using an organic glass hydrophore and stored in two 1-L polyethylene bottles. One bottle was used to identify and count the phytoplankton (Olympus microscope, Japan; Shineso Algacount, China); the contents of this bottle were immediately preserved with 5% formalin and 1% Lugol's solution. The other sample was used to measure the total phosphorus (TP; mg/L), ammonium (NH_4–N; mg/L), iron (Fe; mg/L), and manganese (Mn; mg/L) contents as well as the chemical oxygen demand (COD_{Mn}; mg/L). Unless otherwise stated, all samples were stored at 4 °C immediately after collection, and all of the parameters were measured within 72 h using officially recommended analysis methods (Rice and APHA, 2012).

2.3. The water-lifting aerator in Shibianyu Reservoir

As shown in Fig. 2a, the WLA consisted of an air supply pipe, air-releasing tube, aeration chamber, return chamber, ascending tube, watertight compartment and anchor pier. The WLA was fixed to an anchor pier by non-rust steel wire ropes, and its inlet was located 3 m from the reservoir bed. The air-releasing tube is an annular pipe with a diameter of 80 mm that is perforated with 564 3-mm diameter holes. The compressed air was continuously delivered to the air-releasing tube and then released to the aeration chamber in the form of small bubbles. The aeration chamber and return chamber provide the bubbles with sufficient time to deliver the oxygen to the water. After oxygenation, the bubbles advance the water to the bottom of the reservoir via the return chamber. This function can directly oxygenate the water in the lower layer and circulate the lower, anoxic water, which promotes the diffusion of dissolved oxygen. The off-gas was collected in the air vessel. When

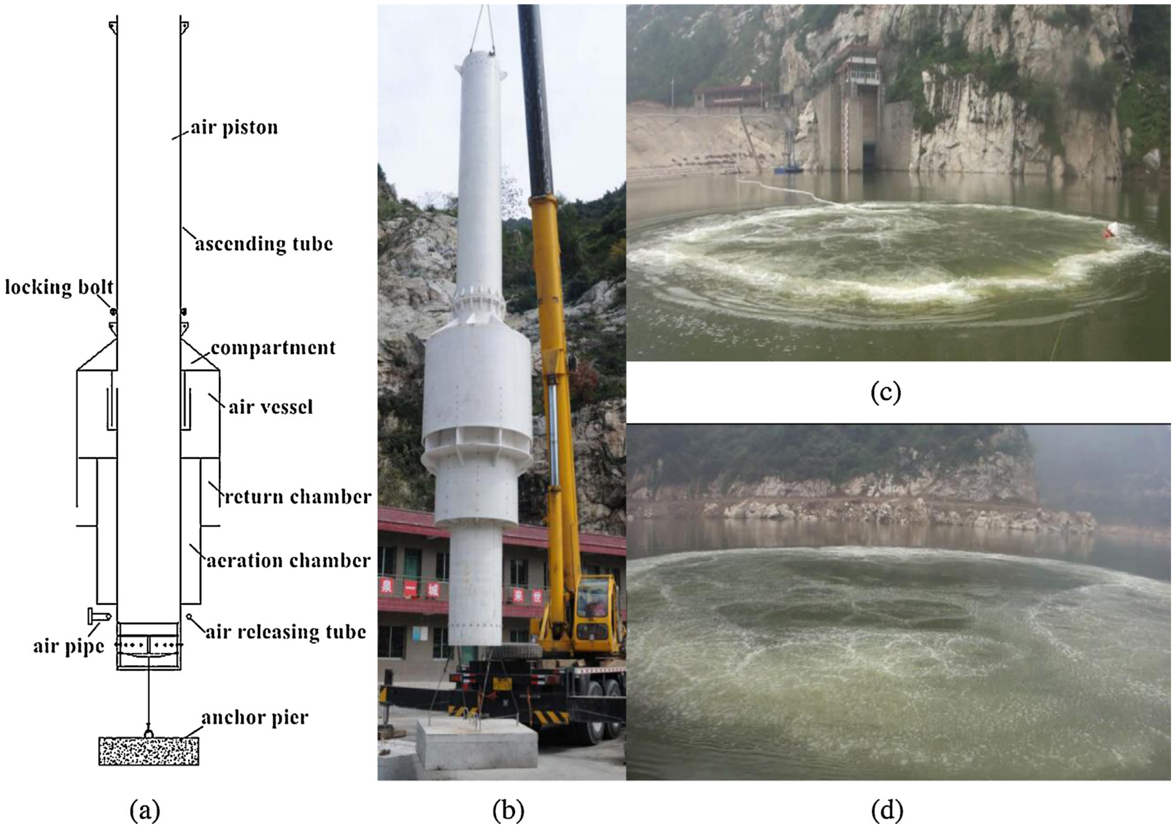

Fig. 2. Diagram of the WLA structure (a), the assembled WLA made from the new FRPP material (b) and the picture when 1# WLA (c) and 2# WLA (d) was in operation.

the air vessel is full, the gas is instantaneously released to the ascending tube. Subsequently, a large air piston forms, rises promptly and then accelerates the water in the ascending tube and part of the water in the lower layer, which continues to rise until the air piston exits the outlet of the ascending tube.

Fig. 2b shows the assembled WLA made from the new fiber reinforced polypropylene (FRPP) material. This material was easy to install and maintain; and it also resists corrosion in water compared with the previously used steel panels. The installation of WLAs was completed in late August 2012. The reservoir experienced heavy rainfall prior to the operation of the WLA. This heavy rain lasted for two days and generated a total rainfall of 134.1 mm. The reservoir level rose by 12.8 m due to this storm runoff, which strongly influenced the stratification structure. The water temperature difference between the surface and the lower layer decreased due to the rainstorm runoff near 1# WLA. 2# WLA was located at the deepest point of the SBYR. After the heavy rain, the water temperature at the bottom remained at approximately 10 °C at the S7 monitoring site, while the lower layer water temperature at the S8 monitoring point increased to approximately 15 °C.

Due to the high nutrient input into the reservoir by the heavy rain, cyanobacteria blooms appeared again approximately one week after the heavy rain. The two water-lifting aerators then began to operate to improve the water quality for this study. Fig. 2c and d shows a photograph of the scene when the WLAs were in operation. The WLAs operated from September 10 to October 22, and the initial 10 days were the debugging stage (the WLAs only operated in the daytime to investigate the stability of the new material). The submerged depth, pressed air flow and time interval of the air piston during regular operation (24 uninterrupted hours) were approximately 15 m, 14 m^3/h and 4.1 min for #1WLA and 35 m, 28 m^3/h, 2.3 min for #2 WLA, respectively.

3. Results and discussion

3.1. Rainfall and hydrological conditions of SBYR

Influenced by the temperate monsoon climate, more than 60% of the annual precipitation at the SBYR was concentrated in the flood season from July to September, with an increasing frequency of extreme rainfall events in recent years. The change in the inflow volume strongly influenced the thermal stratification of the SBYR. As shown in Table 1, 2011 was a wet year with an annual total rainfall of 1072 mm. The total inflow volume in September reached 47.35 million m^3, which was 1.68 times the capacity of the SBYR. In 2012, the annual total rainfall was 782.1 mm. Compared to rainfall in 2011, the monthly rainfall was distributed more homogeneously during the flood seasons of 2012, with 163.1 mm, 180.2 mm and 172.2 mm. The largest peak flow was 117.6 m^3/s (the normal inflow volume is less than 1 m^3/s) at the end of August, 2012. The change in the rainfall and hydrological conditions significantly influenced the water quality of the SBYR, which will be discussed later.

3.2. Water quality problems of SBYR

3.2.1. Hypolimnion anaerobic condition

Fig. 3a and c shows that the thermal stratification in the SBYR started in April and became stable in late May, with a rapid increase in the surface water temperature and a slow change in the temperature of the lower water layers. This structure limited the oxygen transfer from the surface to the lower hypolimnetic layer. As the water and sediment were consumed, the available dissolved oxygen in the hypolimnion rapidly decreased. The hypolimnion became anoxic during the stratified period, which increased the release of pollutants from the bottom sediments (Chen et al., 2015). The temperature stratification in the SBYR was strongest in August and began to diminish in late September. In general, the SBYR was completely mixed in mid-December, when the surface water temperature fell below hypolimnion water temperature. The SBYR was stratified for most of the year, and longer periods of anoxia more significantly deteriorated the water quality as nitrogen, phosphorus, iron and manganese diffuse out of the sediment and entered to the hypolimnion (Huang et al., 2014a). The release of nutrients exacerbated eutrophication, which increased the difficulty and cost of water treatment. Moreover, the increases in the iron and manganese contents deteriorated the water quality by causing or aggravating problems related to water color and odor as well as contributing to acidification (Chiswell and Zaw, 1991)

The high inflow volume induced the early mixing of the SBYR in mid-September, 2011. The water temperature and dissolved oxygen in the hypolimnion simultaneously increased (Fig. 3). The continuous high inflow volume and decreased air temperature retained the reservoir in a mixed state, effectively restraining the release of internal pollutants. The heavy storm runoff at the end of August 2012 increased the hypolimnion water temperature and dissolved oxygen content at site S12 (Fig. 3c). However, the easily degraded organic matter that entered the reservoir via the storm runoff increased the oxygen consumption rate in the water. After the heavy rainfall, the bottom water became anoxic again at site S12 within 20 days (Fig. 3d). The water at site S12 completely mixed in advance at the beginning of November due to the increase in the water temperature at the bottom, which was influenced by the storm runoff (Fig. 3c and 3d). This storm runoff did not affect the lower layer water at site S7 (the deepest site in the SBYR). After the storm runoff, the hypolimnion water remained anaerobic, with a water temperature of approximately 8 °C at site S7. As shown in

Table 1
Monthly rainfall and inflow volume in SBYR form 2011 to 2012.

Month	2011		2012	
	Precipitation (mm)	Inflow volume (10^4 m^3)	Precipitation (mm)	Inflow volume (10^4 m^3)
January	5.4	13	9.8	53
February	12.3	24	0	25
March	21.6	134	16.3	604
April	35.3	445	34.0	409
May	129.3	615	86.5	691
June	57.8	464	22.2	145
July	216.7	828	163.1	804
August	72.4	1691	180.2	642
September	412.0	4735	172.2	1869
October	49.4	888	49.8	154
November	49.0	848	43.0	103
December	10.8	211	5.0	27
Total	1072	10896	782.1	5526

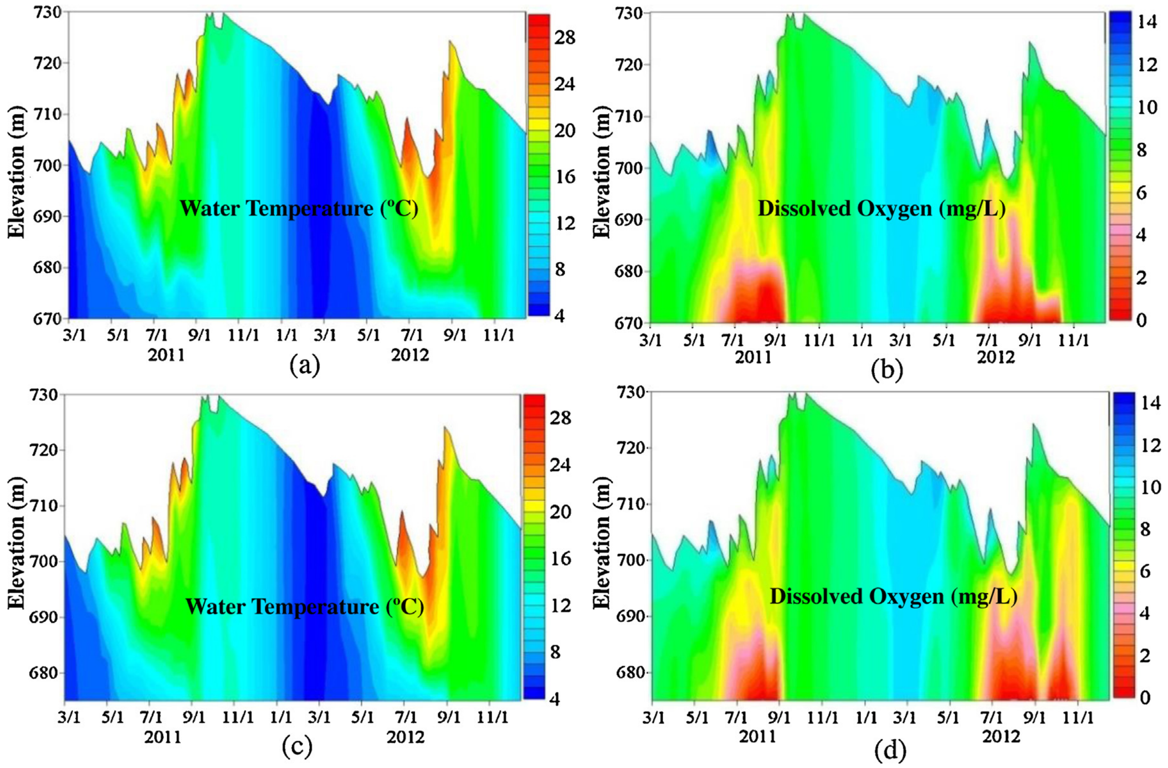

Fig. 3. Vertical variation of water temperature, dissolved oxygen at S7 (a, b) and S12 (c, d) from 2011 to 2012.

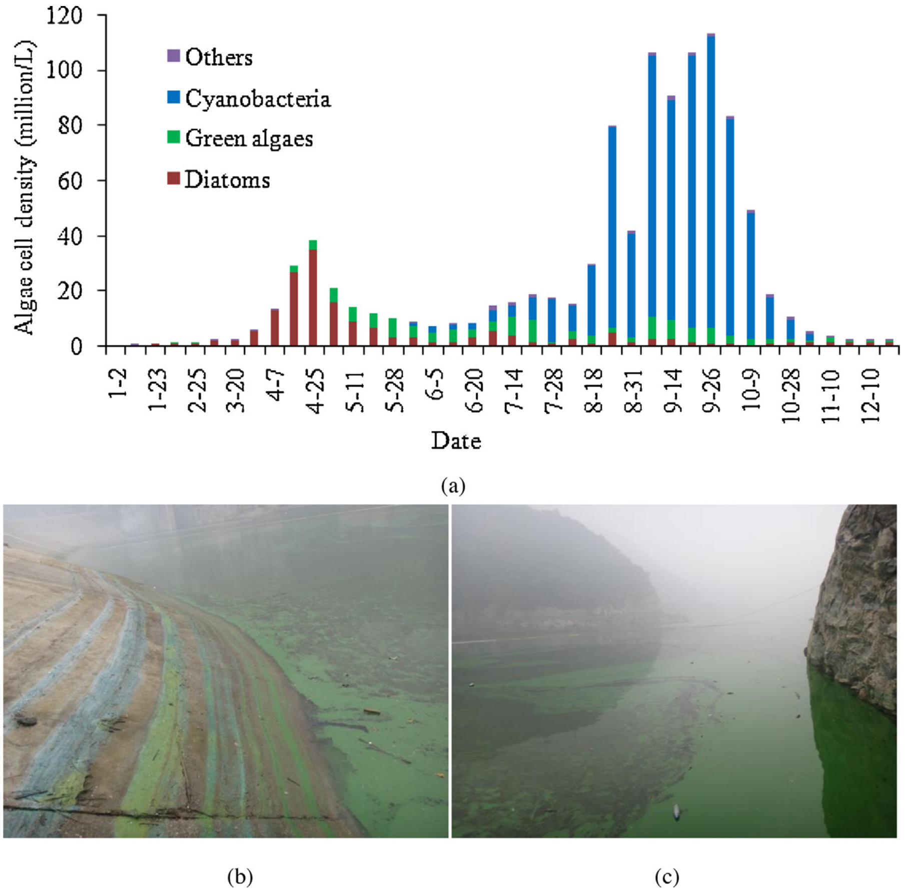

Fig. 4. Seasonal variations in the phytoplankton biomass and community structure (a) and cyanobacteria blooms pictures (b, c) in Shibianyu Reservoir (the images b and c were taken on October 8, 2012).

Fig. 3, S7 was completely mixed before S12 due to the operation of the WLAs, which is discussed in detail below.

3.2.2. Cyanobacterial blooms

Fig. 4 shows the seasonal variations in the phytoplankton biomass and community structure at the S1 monitoring site in 2012, indicating intense phytoplankton blooms in the SBYR. The high nitrogen and phosphorus concentrations of the SBYR provided sufficient nourishment for algae to multiply (Huang et al., 2014a). As the temperature and light intensity increased, algae began to thrive starting in March. The growth of diatoms peaked in the middle of April, and the algal cell density reached 37 million per liter, 87% of which consisted of Fragilaria (Fig. 4a). Green algae predominated in June, but a growth peak was absent due to the competition with other algae (Fig. 4a). Starting in July, sufficient illumination conditions, higher water temperatures and stable water stratification promoted the multiplication of cyanobacteria (Fig. 4a). Cyanobacteria can adjust their buoyancy; therefore, they primarily aggregate between the surface and a depth of 0.3 m. The growth rate of algae is higher at this depth. Intense cyanobacterial blooms have been observed in the SBYR between July and October. A large number of green flocs appeared on the surface of the reservoir, and the algal cell density reached 100 million/L, of which 92% consisted of Microcystis, which seriously affected the ecological balance of the reservoir (Fig. 4c,d).

3.3. Water quality improvement by the water-lifting aerators

3.3.1. Mixing effects

The large air piston formed in the ascending tube promptly ascended and then accelerated the water in the ascending tube, which continued to rise until the air piston exited the outlet of the ascending tube. Subsequently, the bottom water continued to rise under the effect of inertia and moved horizontally due to its inherent kinetic energy upon arrival at the water surface. In a stratified reservoir during summer, the water at the bottom was denser than the surface water. Low-temperature water reached the surface and initially moved horizontally in all directions due to its inherent kinetic energy; the water then moved downward because it was denser than the background water. However, the water changed direction and moved upward because of the buoyancy resistance from the stratification (Huang et al., 2014b).

The temperature gradient is an important factor that affects the mixing depth of the WLA (Sun et al., 2013). As the water temperature gradient decreased, the buoyancy resistance decreased, which increased the mixing depth. The temperature gradient near the WLAs at a water depth of 45 m or less was only 0.1 °C/m, while a large temperature gradient was observed between 45 m and 50 m (approximately 1.3 °C/m) after the storm runoff (Fig. 5c, S7). The water temperature is an important index to measure the mixing effects of the WLAs. As shown in Fig. 5b and d, the water body was completely mixed (water temperature difference between the surface and bottom was less than 0.5 °C) at depths of 40 m or less near the WLAs after the aerator operated for three weeks, while a relatively stable temperature stratification structure persisted within 5 m of the bottom sediment at S7 (the deepest monitoring site). The bottom water at S7 was completely mixed after 6 weeks, which is discussed in the following section. The effective radius of the WLAs was the main factor to determine the installation of WLAs. Many factors influence the effective radius of WLAs, such as the submerged depth, pressed airflow, WLA structure, etc. 1# and 2# WLA were submerged at depths of

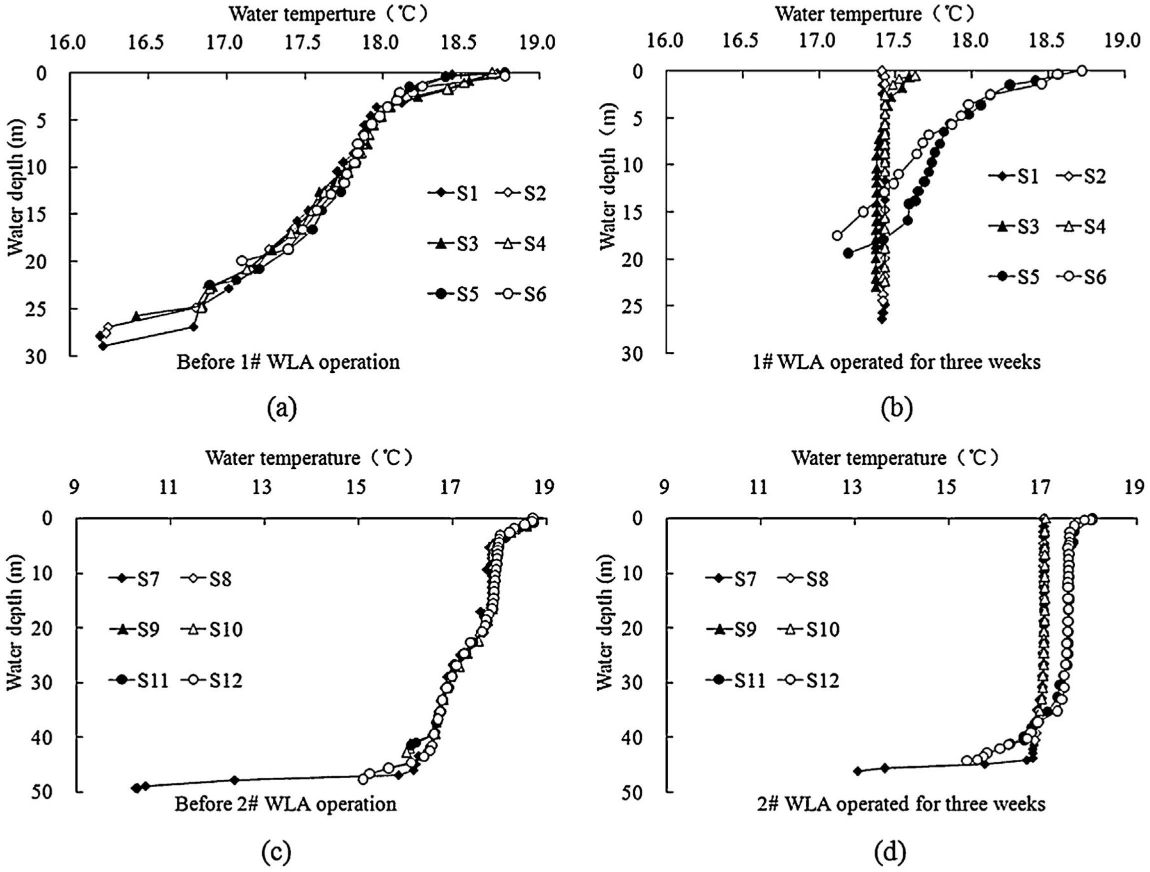

Fig. 5. Vertical distribution of water temperature before (a, c) and after (b, d) the two water-lifting aerators operation at all monitoring sites.

approximately 15 m and 35 m, respectively. The pressed airflows of 1# and 2# WLA were approximately 14 m^3/h and 28 m^3/h, respectively. After operation for three weeks, the radii of the two WLAs reached 60 m and 80 m, respectively (Fig. 5b,d).

3.3.2. Oxygenation of the water in the lower layer

The WLA can increase the concentration of dissolved oxygen of water through in the lower layer in two ways: (1) destroy the water stratification by mixing the lower and upper water layers, which increases the dissolved oxygen concentration in the lower layer; (2) directly oxygenate water in the lower layer and decrease the circulation in the lower, anoxic water, which promotes the diffusion of dissolved oxygen. As shown in Fig. 6a, the water temperature difference was approximately 4 °C within 3 m of the bottom 10 m from the 2# WLA before the operation of the aerator. The mixing function of the WLA continuously increased the water temperature of the lower layer. The water temperature difference in the lower layer water influences the diffusion rate of dissolved oxygen. As shown in Fig. 6b, the dissolved oxygen content increased more slowly near the reservoir bottom at the beginning of the operation of 2# WLA. As the water temperature difference decreased, the diffusion rate of dissolved oxygen significantly increased. After one month of WLA operation, the dissolved oxygen 0.2 m above the reservoir bottom increased from 0 mg/L to 5 mg/L. Due to the rainstorm runoff, the water temperature increased to approximately 16 °C and the oxygen dissolved in the bottom water increased from 0 mg/L to 4.2 m/L at the S12 monitoring point (100 m from the 2# WLA) after the heavy rain. During the operating period of the WLA, the water temperature slightly decreased, while dissolved oxygen content rapidly decreased in the lower layer (Fig. 6c and d).

3.3.3. Effect on inhibiting the release of internal pollutants

Anaerobic conditions can lead to the release of pollutants from the sediment. The WLA can effectively restrain the release of pollutants from the sediment by increasing the dissolved oxygen content in the hypolimnion. As shown in Fig. 7, the concentrations of dissolved phosphorus, ammonia, iron and manganese in the overlying water reached 0.116 mg/L, 0.63 mg/L, 0.51 mg/L, and 0.61 mg/L, respectively, prior to the operation of the WLA. As the dissolved oxygen content at the bottom increased to above 2 mg/L, the release of pollutants from the sediment was effectively suppressed after the operation of the WLA (Jiang et al., 2008). The mixing due to the WLA rapidly decreased the concentration of pollutants concentration in water at the bottom. After the operation of 2# WLA for one month, the concentrations of dissolved phosphorus, nitrogen from ammonia, iron and manganese in the overlying water decreased to 0.048 mg/L, 0.20 mg/L, 0.21 mg/L, and 0.10 mg/L, respectively.

3.3.4. Effect on inhibiting algae growth

According to the distribution and growth characteristics of phytoplankton, the WLA aims to control algal growth by reducing the quantity of algae in the upper layer of the water while increasing it in the lower layer, which homogenizes the algal distribution. The decrease in the algae in the upper layer reduces their production capacity, while the increase in algae in the lower layer accelerates algal death due to the unfavorable growing conditions. As shown in Fig. 8a, the algae were mainly concentrated in the upper layer of the reservoir prior to the operation of 1# WLA.

After the operation of 1# WLA, the quantity of algae in the upper water layer markedly decreased, while the number of algae

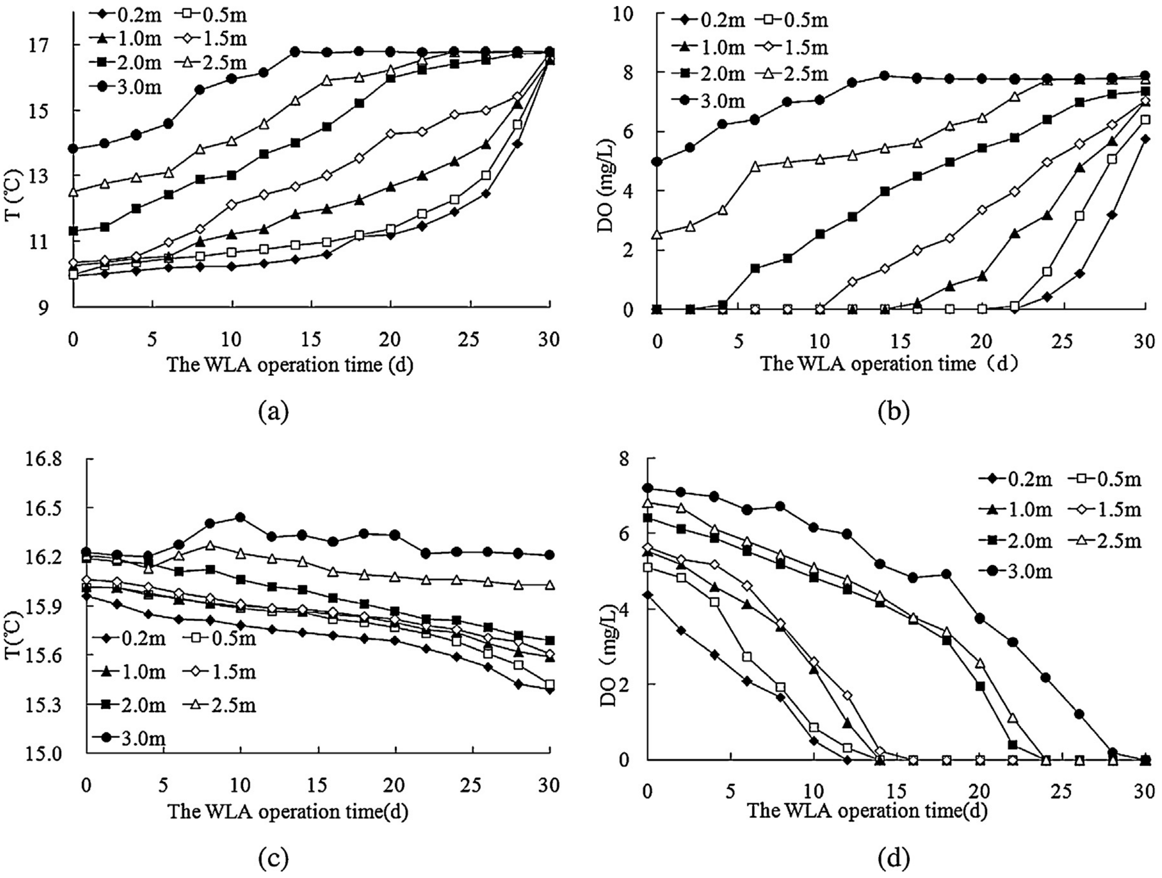

Fig. 6. Changes of water temperature and dissolved oxygen in the deep layer water at S7 (a, b) and S12 (c, d) after the WLA operation.

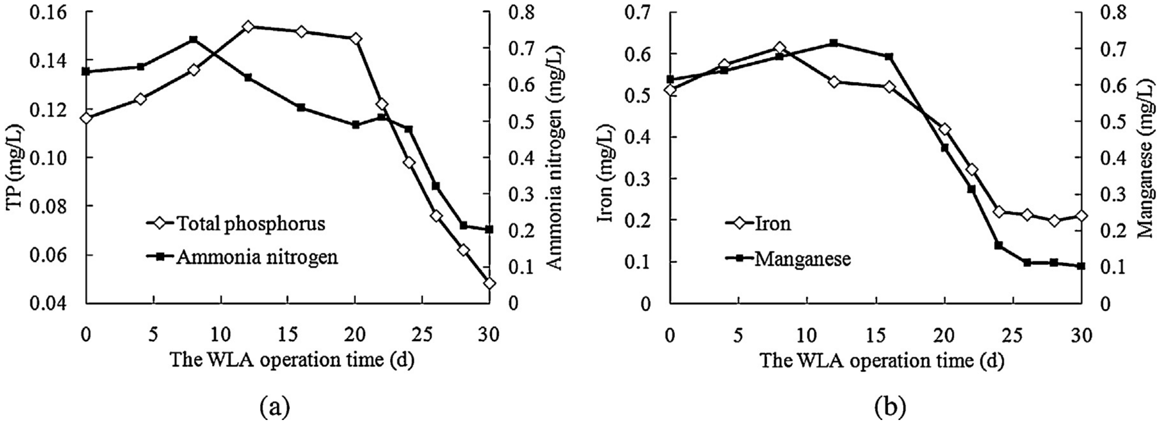

Fig. 7. Dynamic changes of total phosphorus, ammonia nitrogen (a), total iron and manganese (b) at the bottom of S7 after the WLA operation.

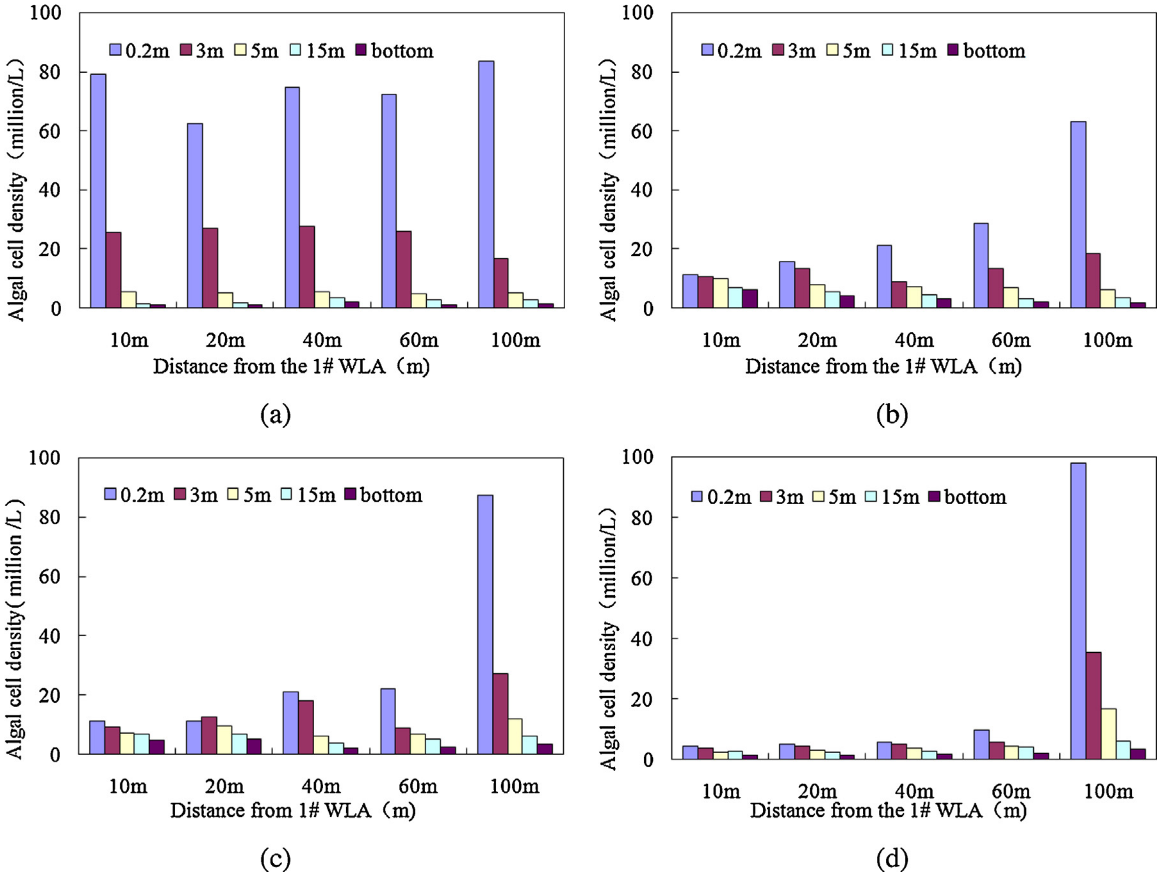

Fig. 8. Vertical distribution of algae at different distance from the aerator before (a) and after the 1# WLA operation for one week (b), two weeks (c) and three weeks (d).

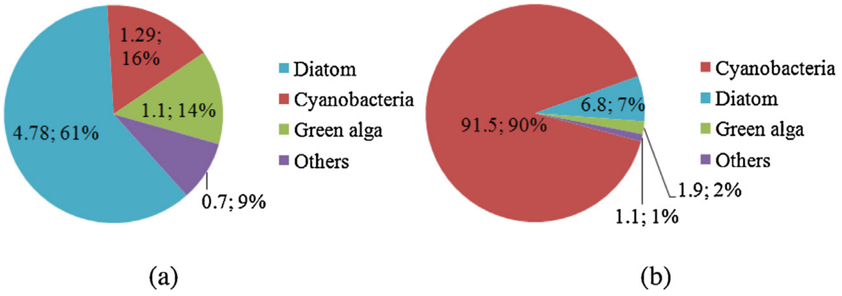

Fig. 9. Phytoplankton community structure and quantity at the surface of S1 (a) and S6 (b) after the aerator operation for three weeks.

at the bottom increased (Fig. 8b,c). The algae growth within 60 m of 1# WLA was effectively suppressed after the operation of 1#WLA for three weeks, while the algae number reached 100 million/L 100 m from 1# WLA.

Cyanobacteria can adjust their buoyancy to exploit nutrients, light and inorganic carbon resources over a wider range. The vertical circulation caused by the operation of the aerator can effectively resist the flotation of cyanobacteria, reduce the harmful competitive advantage of algae, and thus change the structure of the phytoplankton community. As shown in Fig. 9, the algal cell density decreased to less than 10 million/L, of which only 16% consisted of cyanobacteria at S1. The quantity of algae reached 100 million/L, of which 91% consisted of Microcystis at S6 after the aerator had operated for three weeks. The structure of the phytoplankton community was changed same as the situation in Lake Dalbang (Heo and Kim, 2004), Solomon Dam (Hawkins and Griffiths, 1993) by the operation of artificial aeration and in Bleiloch Reservoir (Becker et al., 2006) (even only the upper 20 m of the water column were destratified). Thus, the operation of a WLA can both reduce the number of algae and change the phytoplankton community structure.

3.3.5. Effect on accelerating organic matter degradation

The concentration of organic matter increased because of the large amount of suspended matter that entered the reservoir via the storm runoff. Prior to the operation of WLAs, the concentration of COD_{Mn} reached approximately 7 mg/L in the SBYR. Due to the aerobic biodegradation, the concentration of COD_{Mn} consistently decreased (Table 2). Compared to the sites 100 m away from the WLA, the average organic degradation rate was higher at sites close to the WLA. The results demonstrated that the activity of microbial metabolism was improved, which can accelerate the rate of organic matter degradation to 0.11 mg/L × day after the operation of the WLA.

4. Conclusions

(1) The WLA can increase the dissolved oxygen content of the deep layer water in two ways: by destroying the water stratification by mixing the water in the lower and the upper layers, which increases the dissolved oxygen concentration in the lower layer, and by directly oxygenating the water in the lower layer. After the WLA operated for one month, the dissolved oxygen at 0.2 m above the reservoir bottom increased from 0 mg/L to 5 mg/L.

(2) As the oxygen dissolved in the bottom water increased, the release of pollutants from the sediment was effectively suppressed. After 2# WLA operated for one month, the concentrations of dissolved phosphorus, ammonia nitrogen, iron and manganese in the overlying water decreased to 0.048 mg/L, 0.201 mg/L, 0.201 mg/L, and 0.103 mg/L, respectively.

(3) The mixing function of the WLA can effectively control algal growth by mixing the upper and lower water layers to transport the algae in the upper layer down to the lower layer. The WLA mixed conduction speed gradient can effectively resist blue green algae floatation, reduce the competitive advantage harmful algae, and thus change the structure of the phytoplankton community.

(4) The operation of the WLA can simultaneously improve the activity of microbial metabolism, which can accelerate the rate of organic matter degradation.

Table 2
The concentration and averaged degradation rate of COD_{Mn} during the WLA operation time.

Sites	The WAL operation time (day)							Averaged degradation rate (mg/L × day)
	0	5	10	15	20	25	30	
10 m	6.98	6.12	5.38	4.98	4.25	4.15	3.62	0.1120
20 m	7.18	6.28	5.47	5.11	4.36	4.22	3.78	0.1133
40 m	6.98	6.23	5.52	5.09	4.42	4.32	3.82	0.1053
60 m	7.02	6.15	5.73	5.23	4.57	4.49	3.79	0.1077
100 m	6.89	6.32	6.02	5.67	5.25	5.11	4.92	0.0657

Acknowledgments

This study was supported by the National Natural Science Fund of China (Nos. 51478378, 50830303) and the program of the National Science and Technology Pillar Program (2012BAC04B02). The authors would like to express their gratitude to the agencies involved and to the study participants.

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