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Reducing eutrophication risk of a reservoir by water replacement: a case study of the Qingcaosha reservoir in the Changjiang Estuary

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Abstract
Eutrophication of freshwater systems in cities is a major concern worldwide. Physical, biological and chemical methods have been used in eutrophic lakes and reservoirs to reduce their eutrophic state and algal biomass, but these approaches are not effective without a substantial reduction in nutrients input, which could take decades to achieve in the developing countries. This study aims to assess the risk of eutrophication and algal bloom in a coastal reservoir with high nutrient inputs to confirm the feasibility of inhibiting the reservoir's eutrophic state by hydrodynamic operations. A variety of water quality indexes (e.g., water temperature, secchi depth, dissolved oxygen, total nitrogen, total phosphorus, phytoplankton chlorophyll *a*) at five observed sites were investigated in the Qingcaosha reservoir, which located in the Changjiang Estuary, during the construction, trial and normal operation periods from 2009 to 2012. No water exchange happened during the construction from April 2009 to October 2010, and the water exchange increased during the trial from October 2010 to January 2011, and during normal operation period from January 2011. The comprehensive nutrition state index (*TLI*) calculated by several representative water quality indexes was adopted to evaluate the variation of the trophic state in the reservoir. The peak values of *TLI* reached 51 in the summer of 2009, and 55 in the summer of 2011, higher than the eutrophication threshold value 50. The lowest *TLI*, about 32, appeared in the summer of 2010. The values of *TLI* in other observation periods could keep under 50. The results showed that the reservoir could easily deteriorate into the eutrophic state because of excess nutrients and algal blooms in the summer of 2009 and 2011, while the eutrophication and algal blooms could be reduced by the lack of nutrients in 2010 or adequate water replacement in 2012. The temporal and spatial variations of water quality indexes were presented based on observation data and analysis. The adequate water replacement in the reservoir driven by tides was tested to be an efficient and economical method for controlling eutrophication and algae blooms in the water environment with high nutrient inputs.

Key words: estuarine reservoir, eutrophic state, algal bloom, operation way

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1 Introduction

Aquatic systems impacted by human activities continue to deteriorate with the economic development. Because of the high input of nutrients, many freshwater systems have become eutrophic, and the resulting algae blooms have triggered drinking water crises throughout the world (Carmichael, 2001; Huisman et al., 2005; Paerl et al., 2001). Lake Erie (US/Canada), Lake Winnipeg (Canada), Lake Victoria (the largest of the African rift lakes), and Lakes Biwa and Kasimagaura (Japan's largest lakes) have suffered from eutrophication (Paerl et al., 2011). Many lakes and reservoirs in China, especially those near urban areas, have deteriorated to a eutrophic state due to excessive inflows of nitrogen, phosphorus, organic matter and other pollutants (Chen et al., 2009). The Chinese government and many researchers have adopted measures to recover and manage these eutrophic lakes. Restoration projects have been implemented at the Lake Taihu (Pu et al., 1993), Lake Dianchi (Li et al., 2005), Lake Wuli (Chen et al., 2006), Lake Mochou (Pu et al., 2001), and other small lakes and reservoirs (Tu et al., 2004) throughout China.

As the financial center of China, Shanghai is home to more than twenty million residents. Poor water quality has troubled this city for many years. In 2007, a coastal reservoir project was initiated in the Changjiang Estuary to build the largest drinking water source in Shanghai. This estuarine reservoir, named the Qingcaosha Reservoir, was designed to supply 7 190 000 m^3 of freshwater per day (more than 50% of the total freshwater supply in Shanghai) and serve more than 13 million people in Shanghai. It was finished in 2010 and started normal operation in 2011. It is located on the northern end of Changxing Island (Fig. 1). The distribution of freshwater and saltwater fluctuates due to tidal activity in this area (Chang et al., 2014; Qiu and Zhu, 2013; Wu et al., 2006). Utilizing the difference of water level between inside and outside of the reservoir, the water gates at the northwest (upper reach) and southeast (lower reach) of the Qingcaosha Reservoir

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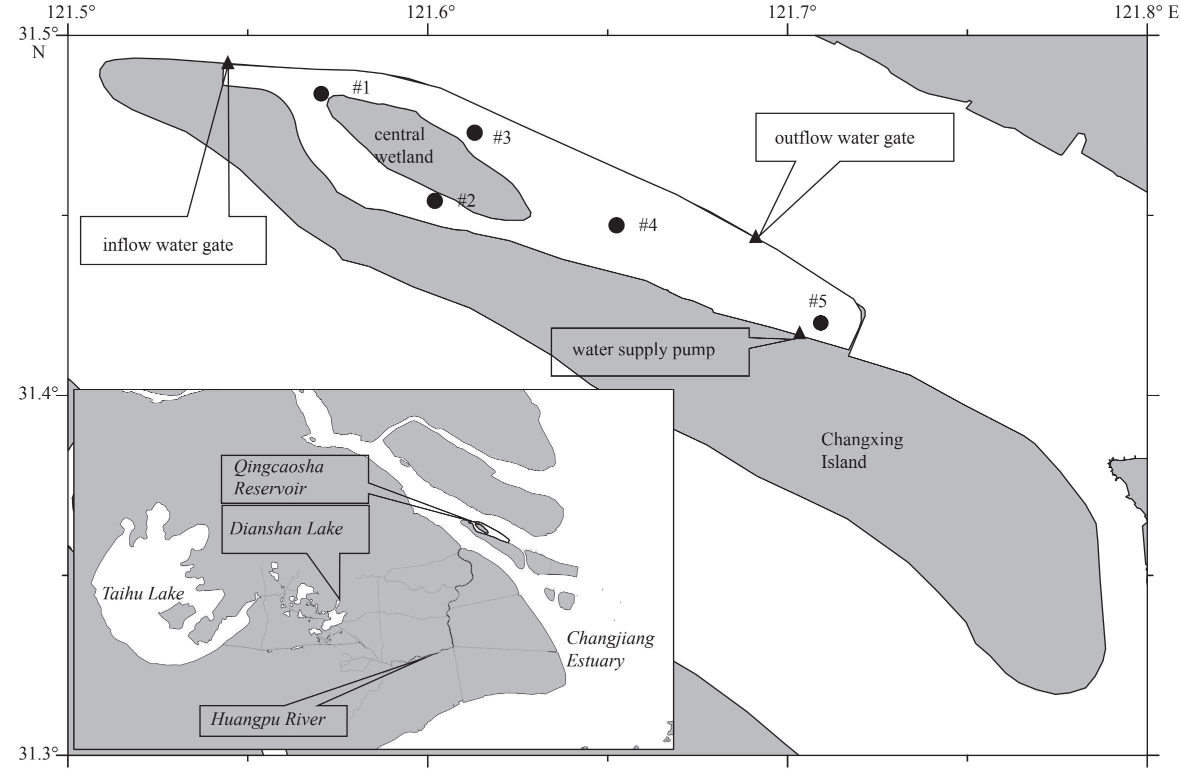

Fig. 1. Map of the Changjiang Estuary and the main fresh water resources near Shanghai. The study region is the Qingcaosha Reservoir which located on the northern end of Changxing Island. The black dots with numbers in the reservoir denote the observation sites.

are manually controlled to draw and discharge freshwater and to avoid saltwater when replacing water in the reservoir. The total area of the reservoir is 66.15 km^2. The water depth in the reservoir varies from 2.7 m to 12.1 m, generally increasing from the northwest (upper reach) to the southeast (lower reach). A large central wetland was set in the northwest central position of the reservoir. This reservoir was enclosed and had no water exchange with the Changjiang Estuary while under construction from April 2009 to September 2010. The project was completed and began to draw water in October 2010. The water supply of the reservoir was approximately 800 000 t/d and the water gates opened only once per day to draw water from the Changjiang Estuary during the trial period from October 2010 to December 2010. After the trial period, the water supply gradually increased to 2.5×10^6 t/d at the end of 2011 and 4×10^6 t/d at the end of 2012. The water gate operation was changed to run twice per day to intake water in the reservoir by means of the semi-diurnal tides.

The substantial water discharge of the Changjiang River exports abundant nutrients into the estuary. The annual nutrients flowing to the sea by the Changjiang River is markedly higher than in other estuaries in China (Shen et al., 1992). Its annual flows of total inorganic nitrogen, phosphate, silicate and nitrate are 8.88×10^6 t, 1.36×10^4 t, 2.04×10^6 t and 6.36×10^6 t, respectively (Gao and Song, 2005). Sharp reduction of such high nutrient inputs for the Qingcaosha Reservoir is difficult and expensive. The researchers and managers of the Qingcaosha Reservoir want to find some efficient and economical methods to keep the trophic state of the drinking water source at a safe level. This study aims to assess the risk of eutrophication and algal blooms in the estuarine reservoir with high nutrient inputs to confirm the feasibility of inhibiting the reservoir's eutrophic state by appropriate reservoir hydraulic operations.

2 Methods

Five observation sites were designed in the Qingcaosha Reservoir (Fig. 1). Site #1 was located in the northwest of the reservoir; Sites #2 and #3 were located at the southern and northern ends of the central wetland, respectively; Site #4 was located near the center of the reservoir; and Site #5 was located at the southeast end of the reservoir. The depths of Sites #1–#5 are 2.7 m, 4.6 m, 8.6 m, 9.7 m, and 10.8 m, respectively. The investigation began in September 2009 at Site #1 and began in April 2009 at the other sites. The monitoring work was completed in December 2012. The water was sampled at a depth of 0.5 m below the surface and 0.5 m above the bottom at each site. Water temperature, Secchi depth (SD) and dissolved oxygen (DO) were measured at all sites. Chemical oxygen demand (COD_{Mn}), ammonia nitrogen (NH_3-N), nitrate nitrogen (NO_3-N), nitrite nitrogen (NO_2-N), total nitrogen (TN), total phosphorus (TP), dissolved total phosphorus (DTP), and phytoplankton chlorophyll *a* (Chl *a*) were sampled at all sites and analyzed in the laboratory. The sampling, measurement and analysis methods all followed the standard procedures recommended by the Ministry of Environmental Protection of the People's Republic of China (Wei, 2002) (Table 1).

The comprehensive nutrition state index (*TLI*) was adopted to evaluate the trophic state of the Qingcaosha Reservoir. Five water quality indexes (Chl *a*, TP, TN, SD and COD_{Mn}) were selected to calculate the *TLI* (Jin and Tu, 1990). The trophic states of lakes and reservoirs are classified into different levels according the *TLI* values (Table 2), which can range from 0 to 100.

The equation to calculate the *TLI* is as follows:

Table 1. Water quality index analysis methods (Wei, 2002)

Index type	Index name	Analysis method
Physical index	water temperature	thermometer method
	dissolved oxygen	electrochemical probe method
	Secchi depth	Secchi disc method
Nutrient index	ammonia nitrogen	Nessler's reagent spectrophotometry
	nitrate nitrogen, nitrite nitrogen	phenol disulfonic acid spectrophotometry molecular absorption spectrophotometry
	total nitrogen	alkaline potassium persulfate digestion-UV spectrophotometry
	total phosphorus	ammonium molybdate spectrophotometry
	dissolved total phosphorus	ammonium molybdate spectrophotometry
	chemical oxygen demand	potassium permanganate index method
Phytoplankton index	Chl *a*	spectrophotometric method

Table 2. Water trophic state classification according to *TLI* (Jin and Tu, 1990)

Trophic state	*TLI*
Oligotrophic state	$TLI<30$
Mesotrophic state	$30\leqslant TLI\leqslant 50$
Eutrophic state	mild eutrophic state: $50<TLI\leqslant 60$
	medium eutrophic state: $60<TLI\leqslant 70$
	hyper-eutrophic state: $TLI>70$

$$TLI=\sum_{j=1}^{m} W_j \times TLI_j, \tag{1}$$

where m is the number of water quality indexes, j is the number of each water quality index, W_j is the weight of each water quality index, and TLI_j is the calculated *TLI* of each water quality index. Based on the Chl *a* value, the value of W_j is calculated as follows:

$$W_j = r_{ij}^2 / \sum_{j=1}^{m} r_{ij}^2, \tag{2}$$

where r_{ij} indicates the correlation coefficient of the water quality index j, calculated according to the reference parameter Chl *a*. r_{ij} is determined according to the calculated results from the 26 major lakes in China (Jin, 1995) (Table 3).

Table 3. The r_{ij} of water quality indexes (Jin and Tu, 1990)

Correlation coefficient	Chl *a*	TP	TN	SD	COD_{Mn}
r_{ij}	1	0.84	0.82	0.83	0.83

The *TLI* of each water quality index is calculated as follows:

$$TLI\,(\text{Chl } a) = 10 \times (2.5 + 1.086 \times \ln \text{Chl } a)\,, \tag{3}$$

$$TLI\,(\text{TP}) = 10 \times (9.436 + 1.624 \times \ln \text{TP})\,, \tag{4}$$

$$TLI\,(\text{TN}) = 10 \times (5.453 + 1.694 \times \ln \text{TN})\,, \tag{5}$$

$$TLI\,(\text{SD}) = 10 \times (5.118 + 1.94 \times \ln \text{SD})\,, \tag{6}$$

$$TLI\,(\text{COD}_{\text{Mn}}) = 10 \times (0.109 + 2.66 \times \ln \text{COD}_{\text{Mn}})\,. \tag{7}$$

3 Results and discussion

The observation data were analyzed to show the temporal and spatial variations of water quality indexes in the Qingcaosha Reservoir. The calculated result of *TLI* was used to evaluate the trophic state of this reservoir during the construction, trial and normal operation period.

3.1 *Water quality index analysis in the Qingcaosha Reservoir*

There was little spatial variation of water temperature in the reservoir during the observation period. No thermocline was observed in this reservoir. The water temperature in the entire reservoir fluctuated seasonally, ranging from 2.5°C in winter to 32.5°C in summer.

When the reservoir was enclosed, the vertical DO concentrations varied greatly at the measured sites due to the deficiency of water exchange and the influence of phytoplankton photosynthesis at the surface layer. The maximum difference of DO between the surface layer and bottom layer was 6 g/m^3. This phenomenon disappeared after the reservoir began operations. DO also fluctuated seasonally, ranging from 6 g/m^3 in summer to 14 g/m^3 in winter.

The SD in the reservoir was mainly influenced by the growth of phytoplankton and wave induced by wind. The SD generally decreased in summer and increased in winter and spring. The peak value of SD at Site #5 was 1.8 m in the spring of 2010, 1.6 m in the spring of 2011 and 1.7 m in the winter of 2011. The valley value at Site #1 was 0.5 m in the summer of 2009 and 0.17 m in the summer of 2010. With the increasing water exchange in the reservoir, the SD at Site #1 gradually stabilized at 0.25 m. However, obvious fluctuations of SD continued at the other observation sites (Fig. 2). The horizontal variation of SD was primarily increasing from the northwest (the upper reach) to the southeast (the lower reach). This phenomenon could be explained by the settlement process of the suspended solids (SS). The SS was drawn from the northwest gate and kept settling with the water movement to the southeast end of the reservoir.

Based on the annual averaged data at the observation sites, the dissolved inorganic nitrogen (DIN), considered as the sum of NH_3-N, NO_3-N and NO_2-N, accounted for 63.1%–67.8% of the TN. The proportion of particulate nitrogen (PN) in the TN was low. Thus, the TN was not mainly influenced by the wave and SS, and varied little in the vertical during the observation period (Fig. 3). The spatial difference of TN was small during the reservoir's closed period, except that the TN at Site #1 was slightly higher than at the other sites in the summer of 2009 and the spring of 2010, because of the growth of phytoplankton. The concentration of TN in the entire reservoir decreased from 1.8 g/m^3 to 0.5 g/m^3 during the closed period due to the lack of nutrients input and water self-purification (Fig. 2). The concentration of TN at Site #1 clearly increased after October 2010 because of the high nutrient inputs from the Changjiang Estuary. It exceeded 2.6 g/m^3 in July 2011 and fluctuated in the range of 1.2–2.2 g/m^3 in the remaining observation period. During the operation period, the concentration of TN at Site #1 was substantially higher than the other sites, and the others did not obviously differ from each other (Fig. 2). This showed that the TN decreased mainly in the

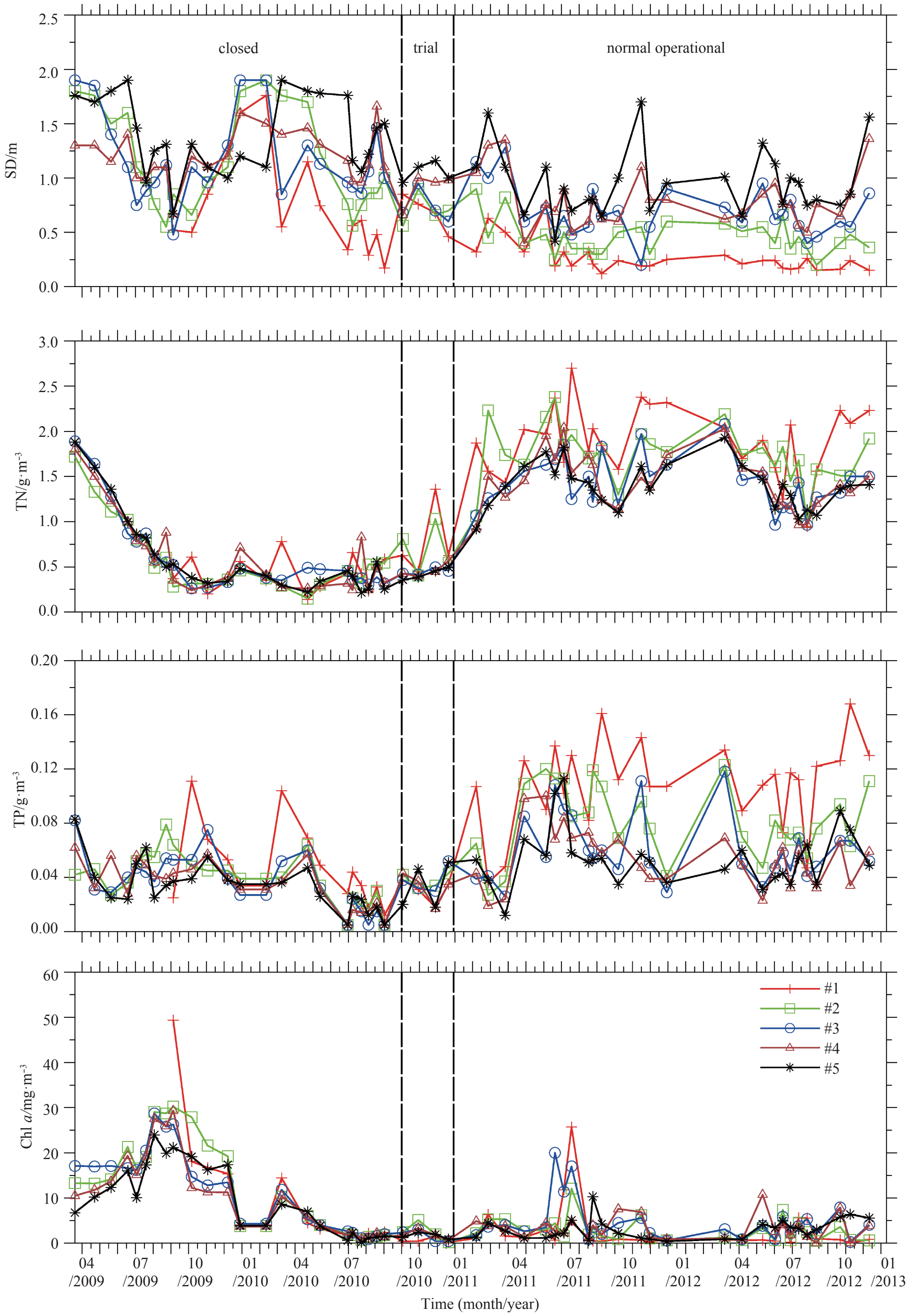

Fig. 2. Time series of SD, TN, TP and Chl *a* from 2009 to 2012 at the observation sites. The values of TN, TP and Chl *a* were observed at a depth of 0.5 m below the water surface. Two vertical dotted lines separate the observation period into three sections: closed period, trial period and normal operation period.

northwest area, because of the nutrients absorption function of the central wetland. Wetlands are well known as the highly effective systems mitigating the negative effects of nitrogen and phosphorus excess. Natural and artificial wetlands are widely applied in the worldwide (Alongi, 2008; Álvarez-Rogel et al., 2016; Kang et al., 2017).

The major component of TP in the reservoir was particulate phosphorus (PP). The annual averaged ratios of DTP/TP were only 26.2%–30.6% at the measured sites. Therefore, an intense fluctuation was found in the analyzed results of TP. The range of TP was 0.005–0.11 g/m^3 during the reservoir's closed period, and this range changed to 0.01–0.17 g/m^3 after the reservoir began to operate (Fig. 2). Due to the settlement of SS, the TP in the surface layer was lower than in the bottom layer from 2009 to 2011. After 2012, the surface concentration of TP was occasionally higher than the bottom concentration (Fig. 3) because of the gradually intensified vertical turbulence during the operation period. The horizontal variation of TP was mainly decreasing from northwest to southeast. Along with the increase of nutrients input, the horizontal gradient of TP gradually increased after the reservoir began to operate in October 2010. This phenomenon was observed in the entire reservoir (Fig. 2), primarily due to the nutrients absorption function of the central wetland and the settlement of PP in the entire reservoir.

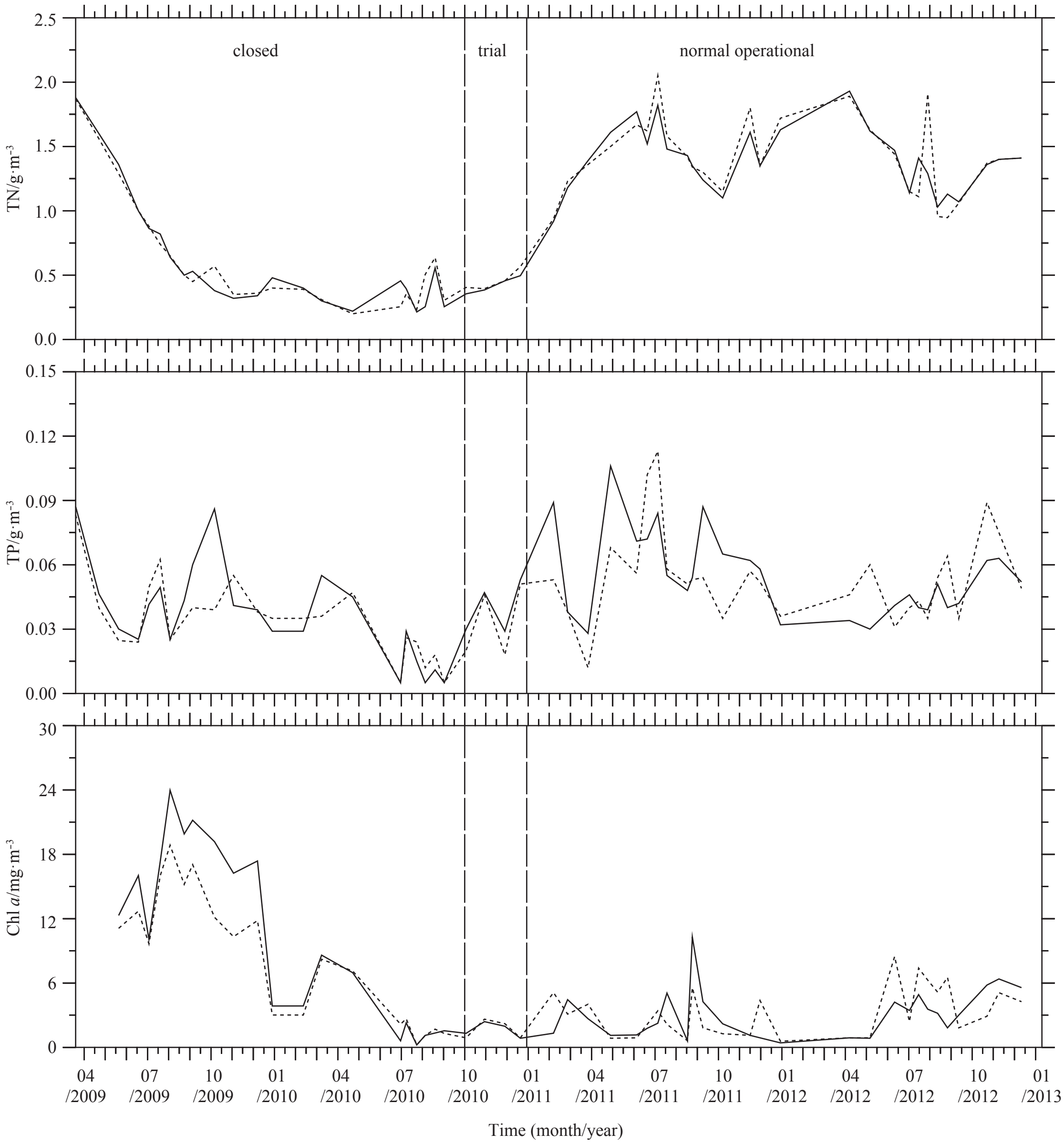

Fig. 3. Time series of TN, TP and Chl *a* from 2009 to 2012 in the different layers of Site #5. The solid line represents surface layer and dashed line bottom layer.

The surface concentration of Chl *a* was much higher than the bottom concentration in the summers of 2009 and 2011, because of the rapid growth of phytoplankton. However, the bottom concentration of Chl *a* was occasionally higher than the surface concentration in 2012 due to the gradually intensified vertical turbulence (Fig. 3). For the stagnant water environment and sufficient nutrients, the phytoplankton grew quickly throughout the reservoir in 2009. An algal bloom occurred and the peak concentration of Chl *a* at Site #1 reached 50 mg/m^3 in the summer of 2009 (Fig. 2). The concentrations of TN and TP decreased to 0.5 g/m^3 and 0.01 g/m^3 in the summer of 2010 because of no nutrient inputs (the reservoir was enclosed) and water self-purification. The Qingcaosha Reservoir had not reached a eutrophic state (Nürnberg, 1996), and the low nutrients could not support the growth of phytoplankton. Therefore, the concentration of Chl *a* was reduced to 2 mg/m^3 and no algal bloom happened in the summer of 2010. After October 2010, the reservoir began to operate, and the fresh water with high nutrients in the Changjiang Estuary was drawn into the reservoir. The growth of phytoplankton rose again and the peak concentration of Chl *a* at Site #1 reached 25 mg/m^3 in the summer of 2011. However, the growth of phytoplankton had been weakened with the gradual increase of water replacement in the reservoir since 2012. The Chl *a* in the entire reservoir was limited to less than 10 mg/m^3 in the summer of 2012, even with high concentrations of nutrients. The variations of Chl *a* in 2012 show that the adequate water replacement is an effective method for controlling algae blooms in a eutrophic water environment. The similar conclusion was reached in the study of Dianshan Lake in Shanghai, the accumulation and growth of phytoplankton in lakes and reservoirs with high nutrients could be effectively inhibited by the increasing discharges (Chen et al., 2016). The main reason of this phenomenon is that the algae can be quickly transported out of lakes and reservoirs by adding inflow and outflow discharges before they have the opportunity to bloom.

3.2 *Trophic state assessment of the water environment in the Qingcaosha Reservoir*

Using the spatially averaged data, the *TLI* was calculated to reflect the variation of the water trophic state in the Qingcaosha Reservoir (Fig. 4). The trophic state of the reservoir deteriorated from mesotrophic state to mild eutrophic state because of the rapid growth of phytoplankton, sufficient nutrients and the obvious decline of SD from July 2009 to October 2009. The trophic state quickly recovered to mesotrophic state due to the low concentrations of nutrients and the slow growth of phytoplankton from November 2009 to September 2010. The reservoir began to operate in October 2010, and the growth of phytoplankton was then reactivated along with the high nutrient inputs from the Changjiang Estuary. Thus, the trophic state of the Qingcaosha Reservoir gradually deteriorated to a mild eutrophic state again in July 2011. With regard to the increase of TN and TP, the peak value of *TLI* in the summer of 2011 was higher than in the summer of 2009. In 2012, the concentrations of nutrients in the reservoir remained high due to the increasing inflow from the Changjiang Estuary, but the growth of phytoplankton was restricted because of the more frequent water replacement (Fig. 2). Thus, the trophic state of the Qingcaosha Reservoir remained mesotrophic in 2012 (Fig. 4).

According the analysis above, the nutrients in the Qingcaosha Reservoir, which is the cause of the eutrophication and algal blooms, are difficult to reduce because the nutrients input from the Changjiang Estuary is high. Reducing the input of nutrients is still the ultimate method to address eutrophication in lakes and reservoirs (Edmondson, 1970; Schindler, 2006), but this could take decades to achieve in the developing countries (Paerl et al., 2011; Qin et al., 2015). The toxins and odorous compounds generated by algae blooms are much more dangerous than normal nutrients in a drinking water system (Burgos et al., 2014; Ma et al., 2013; Song et al., 2007; Ueno et al., 1996), and the technological requirements and costs required to remove the former are much higher than the latter. Biological methods such as filter-feeding fish, shellfish and aquatic plants, and physical methods including mechanical measures and artificial isolation equipment have been used in eutrophic lakes to reduce algal biomass, but these approaches are not effective without a substantial reduction in nutrients input (Chen et al., 2009; Pu et al., 1993). The positive effects of artificial water diversion and exchange have been proven to decrease the concentrations of phytoplankton in non-tidal areas, but it is difficult to exchange water sufficiently over an entire area and the cost is prohibitive (Hu et al., 2008; Li et al., 2013).

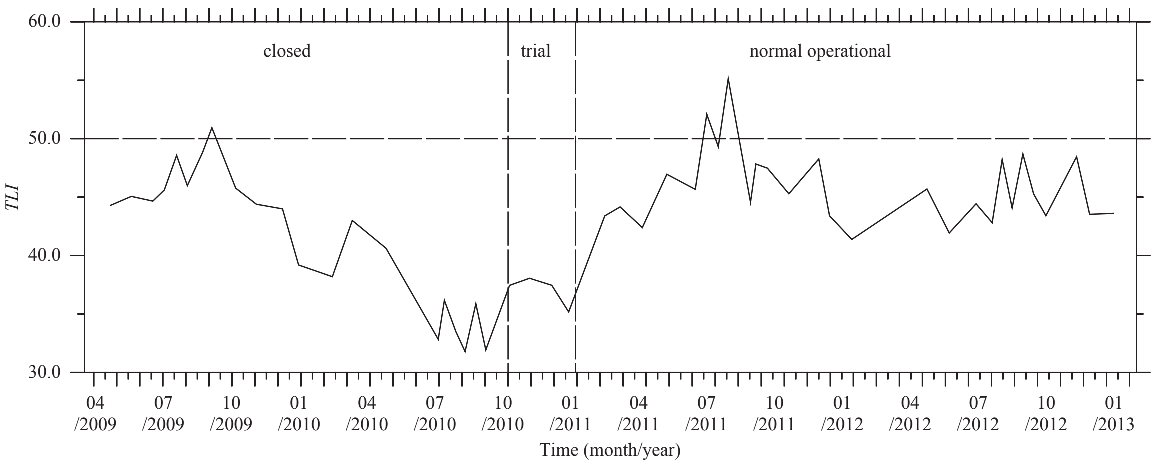

Fig. 4. Time series of *TLI* calculated with the spatially averaged observation data from 2009 to 2012 in the Qingcaosha Reservoir. The horizontal dotted line represents the threshold of eutrophic state according to *TLI*.

The results of this study show that the adequate water replacement driven by tides is an appropriate operation way to inhibit the eutrophic state of estuarine reservoirs with high nutrient inputs. Reservoirs set in estuarine area have a natural advantage in restricting the growth of phytoplankton in the high nutrients environment by utilizing the tidal range, and these reservoirs can exchange water frequently with the control of water gates. It is an efficient and economical way to low down the risk of eutrophication and algal blooms in the drinking water sources, while it takes decades to reduce the input of nutrients loads from the upper reaches to a safe level.

4 Conclusions

The variations of the nutrients and Chl *a* in the Qingcaosha Reservoir were analyzed based on the observed data at five sites from 2009 to 2012. In the summer of 2009, the unfinished reservoir was not connected to the estuary, and the phytoplankton quickly grew because of enough nutrients and the stagnant water condition. In the summer of 2010, the nutrients in the reservoir clearly decreased and no algae bloom occurred due to the lack of nutrients input and water self-purification. The concentrations of nutrients in the reservoir quickly increased and the growth of phytoplankton was reactivated after the reservoir began to operate in October 2010. The peak concentration of Chl *a* reached 25 mg/m^3 in the summer of 2011, but it was limited to below 10 mg/m^3 in 2012 due to the adequate water replacement in the reservoir. The analysis and discussion results suggest that the adequate water replacement driven by tides could be an efficient and economical method for controlling eutrophication and algae blooms in the water environment with high nutrient inputs. The effective inhibition on eutrophication and algae blooms could also decrease the risk of the toxins and odorous compounds generated by algae blooms in the drinking water supplies.

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