

Long-term nutrient trends and harmful cyanobacterial bloom potential in hypertrophic Lake Taihu, China

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Abstract Rapid economic development in China's Lake Taihu basin during the past four decades has accelerated nitrogen (N) and phosphorus (P) loadings to the lake. This has caused a shift from mesotrophic to hypertrophic conditions, symptomized by harmful cyanobacterial blooms (CyanoHABs). The relationships between phytoplankton biomass as chlorophyll *a* (Chl*a*) and nutrients as total nitrogen (TN) and total phosphorus (TP) were analyzed using historical data from 1992 to 2012 to link the response of CyanoHAB potential to long-term nutrient changes. Over the twenty year study period, annual mean Chl*a* showed significantly positive correlations with both annual mean TN and TP ($P < 0.001$), reflecting a strong phytoplankton biomass response to changes in nutrient inputs to the lake. However, phytoplankton biomass responded slowly to annual changes in TN after 2002. There was not a well-defined or significant

relationship between spring TN and summertime Chl*a*. The loss of a significant fraction of spring N loading due to denitrification likely weakened this relationship. Bioavailability of both N and P during the summer plays a key role in sustaining cyanobacterial blooms. The frequency of occurrence of bloom level Chl*a* ($>20 \mu\text{g L}^{-1}$) was compared to TN and TP to determine nutrient-bloom thresholds. A decline in bloom risk is expected if TN remains below 1.0 mg L^{-1} and TP below 0.08 mg L^{-1} .

Keywords Eutrophication · Lake Taihu · *Microcystis* · Chlorophyll *a* · Nitrogen · Phosphorus

Introduction

Eutrophication is a natural process in aquatic ecosystems (Wetzel, 2001; Vieira et al., 2005). However, this process has been accelerated by increased anthropogenic nutrient inputs that have occurred worldwide over the past several decades (Schindler & Vallentyne, 2008; Conley et al., 2009; Qin et al., 2007, 2010). Harmful cyanobacterial blooms (CyanoHABs) are a troubling indicator of eutrophication and are expanding globally (Brookes & Carey, 2011; Paerl & Otten, 2013; Paerl, 2014). These blooms represent a serious threat to water and habitat quality, drinking water supplies, food webs, and the sustainability of freshwater ecosystems (Carmichael, 2001; Huisman et al.,

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2005; Paerl et al., 2011; Paerl & Otten, 2013). CyanoHAB dynamics rely on the synergistic effects of environmental factors, such as nutrients, temperature, light and hydrodynamics, and physiological capabilities of cyanobacteria, such as buoyancy regulation, N₂ fixation, and nutrient storage (Smith, 1990; Dokulil & Teubner, 2000; Zhang et al., 2006; Paerl & Otten, 2013; Paerl and Huisman, 2009). From a management perspective, regulating nutrient inputs is often the most realistic and effective long-term bloom control strategy (Paerl et al., 2016).

Nitrogen (N) and phosphorus (P) are the key nutrients controlling CyanoHAB expansion and persistence (Likens, 1972; Schindler, 1977; Paerl, 2008; Carstensen et al., 2007; Philips et al., 2007; Paerl, 2009; Paerl et al., 2011). Inputs of both nutrients to natural waters have increased dramatically since the 1960s. Reversing eutrophication and reducing CyanoHABs in a range of lakes involves the reduction of P inputs (Schindler, 1977; Schindler et al., 2008), but increasingly reductions of both N and P inputs are required (Kronvang et al., 2005; Elser et al., 2007; Jeppesen et al., 2007; Lewis et al., 2011; Paerl et al., 2015, 2016). Appropriate nutrient reduction strategies are likely to be system specific (Paerl et al., 2016). The central questions facing water researchers and managers are as follows: which nutrient(s) control CyanoHAB production and proliferation, and what nutrient threshold(s) are needed to control CyanoHABs in impacted systems? The answers to these questions are of immense ecological and economic importance, because they dictate the strategies and costs involved in mitigating this serious water quality problem (Paerl et al., 2011, 2016).

Lake Taihu is the third largest freshwater lake in China and is the primary drinking water source for the ~11 million residents of the rapidly expanding cities in the Taihu drainage basin, including Wuxi, Suzhou, Changzhou, and Jiaxing. The Taihu Basin is one of the most industrialized and urbanized regions in China, containing only 0.4% of China's land area, while accounting for 11% of China's gross domestic product (Qin et al., 2007). Rapid economic and population growth in the Taihu Basin has led to a substantial increase in pollution from industries and wastewater treatment facilities that discharge into the numerous tributaries of Taihu. This has accelerated degradation of lake water quality, characterized by increasingly severe and frequent CyanoHABs (Guo, 2007; Qin

et al., 2007; Duan et al., 2009). These blooms have led to serious environmental, economic, and societal problems; for example, the highly publicized drinking water crisis in Wuxi in 2007 (Guo, 2007; Qin et al., 2010). This has added to the urgency for researchers and government agencies to seek mitigation strategies.

The objective of this study was to address whether there is an interpretable and applicable relationship between trends of nutrient concentrations and changes in CyanoHAB magnitude and frequency in Taihu. Using long-term monitoring data of nutrient and biological variables during 1992–2012, correlations between annual average and seasonal average nutrient concentrations (TN and TP), and harmful algal biomass (*Chla*) were analyzed to explore how the changes in N and P concentrations might be related to CyanoHAB dynamics. Nutrient control targets were established based on relationships between nutrient concentrations (TN and TP) and frequencies of CyanoHAB biomass as *Chla*.

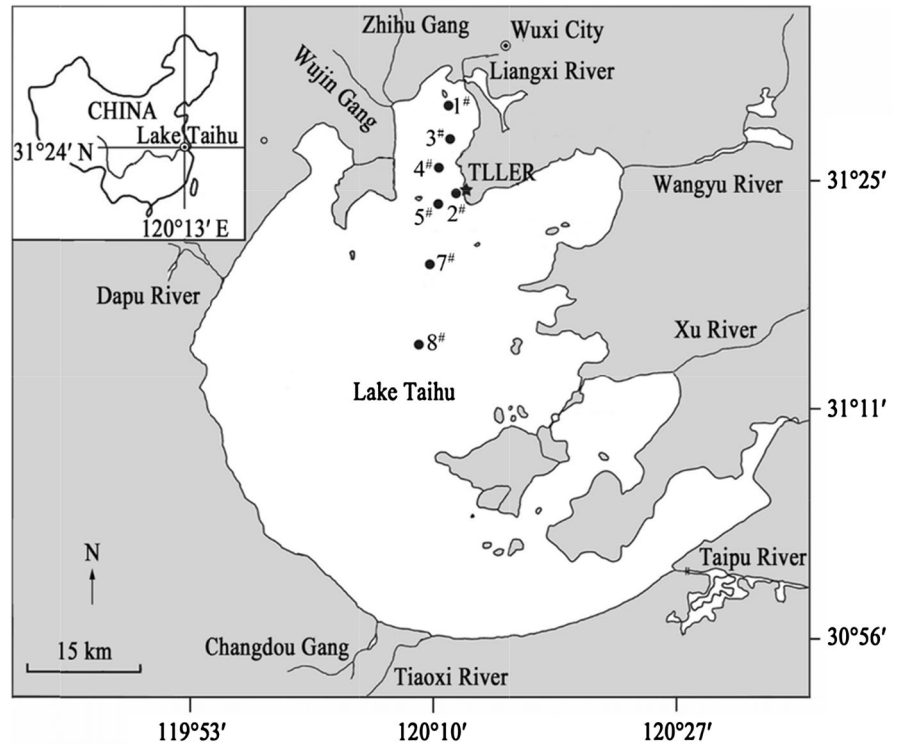
Materials and methods

Location and field sites

Taihu is located in the southeastern region of the Yangtze River delta (latitude 30°55'40"–31°32'58"N; longitude 119°52'32"–120°36'10"E). It is a large, shallow (mean depth ~2 m) polymictic lake, with an area of 2,340 km², a catchment area of 36,500 km² and a volume of 4.4 billion m³ (Qin et al., 2007, 2010). The Taihu basin is characterized by a complex set of river networks, with 117 rivers and tributaries draining into the lake. The annual freshwater input to the lake is about 88 × 10⁸ m³, and the water retention time of the lake is approximately 284 days (Qin et al., 2007). Generally, freshwater inputs enter the lake from the southwest and western mountainous watershed, and the dominant outflows are through Eastern Taihu Bay and the Taipu River, which flows through Shanghai into the East China Sea (Fig. 1).

Meiliang Bay is situated in the northern part of Taihu, and it is one of the most eutrophic regions of the lake. The surface area of the bay is 132 km² with a mean depth of 2.0 m. There are three main inflowing rivers, the Wujing Gang, Zhihug Gang, and Liangxi River, connected to the Meiliang Bay. The catchment of the Wujing Gang and Zhihu Gang contains

Fig. 1 Map of Taihu showing sampling locations in Meiliang Bay (sites 1–5) and central lake (sites 7–8) from which water samples were routinely collected



substantial industry. The catchment of the Liangxi River is less industrial, but the river passes through the city of Wuxi, which has a population over six million (Zhu, 1994).

Data acquisitions and analyses

Water quality and phytoplankton biomass data at five sites inside Meiliang Bay and two sites in the central lake (Fig. 1) were provided by Taihu Laboratory for Lake Ecosystem Research (TLLER.). Water sampling and environmental measurements were performed monthly for the calendar years 1992–2012. At each sampling site, depth-integrated water samples were collected with a 2-m-long, 10-cm-diameter plastic tube. Water samples were analyzed for chemical and biological parameters. Chemical parameters included total nitrogen (TN), total dissolved nitrogen (TDN), ammonium ($\text{NH}_4\text{-N}$), nitrate ($\text{NO}_3\text{-N}$), nitrite ($\text{NO}_2\text{-N}$), total phosphorus (TP), total dissolved phosphorus (TDP), and soluble reactive phosphorus (SRP). SRP was determined using the molybdenum blue method (APHA, 1995). $\text{NH}_4\text{-N}$ was measured by the indophenol blue method, and $\text{NO}_3\text{-N}$ and $\text{NO}_2\text{-N}$ with the cadmium reduction method (APHA, 1995). TP, TDP,

TN, and TDN were analyzed using a combined persulfate digestion (Ebina et al., 1983), followed by spectrophotometric analysis as for SRP and $\text{NO}_3\text{-N}$. TN and TP recovery efficiencies were 98.4 and 99.7%, respectively. Biological parameters included phytoplankton biomass, species composition, and Chl a . Phytoplankton samples were preserved with Lugol's iodine solution (2% final conc.) and sedimented for 48 h. Cell density was microscopically determined using a Sedgwick–Rafter counting chamber at magnifications of 320–400. Phytoplankton species were identified according to Hu et al. (1980). Biovolumes were calculated from cell numbers and cell size measurements. Conversion to biomass was based on 1 mm^3 of volume being equivalent to 1 mg of fresh weight biomass. Chl a concentrations were determined spectrophotometrically after extraction in 90% hot ethanol (Papista et al., 2002).

Data processing and statistical analysis

Monthly nutrient and Chl a data were summarized as mean values for spring (March, April, and May), summer (June, July, and August), autumn (September, October, and November), winter (December, January,

and February), and annual mean values from 1992 to 2012. Annual nutrient-*Chla* relationships were derived using linear least squares regression to examine the effects of annual changes in nutrients on phytoplankton biomass. To explore seasonal changes in the relationship between nutrients and *Chla*, the spring and summer relationships between *Chla* and TN (or TP), as well as the relationship between summer *Chla* and spring TN (or TP), were analyzed using least squares regression. Prior to linear regression analyses, total N, total P, and *Chla* data were log₁₀ transformed to improve normality and homogeneity of variance. Analysis of covariance with time as the covariate was used to determine whether there was a significant interaction between time and nutrient concentration on *Chla*. When significant interaction terms were determined, Tukey–Kramer post hoc tests were used to determine significantly different slopes between the periods.

Chla values exceeding 20 µg L⁻¹ were identified as nuisance bloom conditions based on the summertime observations in Taihu (Xu et al., 2015). Bloom frequency was defined as the percentage of *Chla* observations exceeding 20 µg L⁻¹. Correlations between median concentrations of TN (or TP) and bloom frequency were analyzed to establish bloom nutrient thresholds and control targets. Monthly measurements from the five sampling sites in Meiliang Bay during the bloom season (June–October) from 1992 to 2012 were used for the bloom frequency analysis. Following Havens (2003), the paired total nutrient (N or P) and *Chla* values during the bloom period were sorted based on increasing nutrient values and divided into sixteen approximately equal bins of thirty observations. Median total nutrient values were compared to the percentage of *Chla* within each bin that exceeded 20 µg L⁻¹. Resultant hyperbolic relationships were fit to a log-linear model [$\text{Chla}_{20} = a + b \times \ln(\text{TN or TP})$] as a means of objectively describing the relationships and for assessing uncertainty in the relationship. A best fit model was used to calculate the nutrient level at which the frequency of *Chla* exceeding 20 µg L⁻¹ reached fifty percent. Confidence intervals for these threshold nutrient levels were estimated by standard bootstrapping (Hall & van den Boogaard, 2004). Curve fitting to the log-linear function and estimation of the threshold nutrient level was repeated with 1,000 resampled datasets, each of the same size as the original. The upper and lower

bounds for the 95% confidence interval on the threshold nutrient value were then defined as the 26th and 975th estimate of the sorted threshold bootstrapped estimates (Hall & Boogaard, 2004).

Results

Temporal trends of nutrients and cyanobacterial blooms

Long-term trends of TN, TP, and *Chla* in Meiliang Bay, and the central lake from 1992 to 2012 are shown in Fig. 2. Annual average TN and TP concentrations increased rapidly from 1992 to 1996 and then decreased between 1996 and 2001 in both regions. After 2001, TN and TP concentrations increased rapidly again until 2006, when TN and TP concentrations began to decrease again and reached 1992 levels by 2012. The central lake was less affected than Meiliang Bay by riverine nutrient inputs, as shown by lower TN and TP concentrations. Seasonally, TN in both regions always showed higher concentrations in spring and winter, and lower concentrations in summer and fall during the study period (Fig. 2A, B). From 1992 to 1997, TP concentrations in Meiliang Bay showed highest values in winter or spring. However, from 2000 to 2012, TP concentrations were higher in summer and fall than in winter and spring (Fig. 2C). This trend was not observed in the central lake (Fig. 2D).

Annual average *Chla* concentrations throughout the water column increased from 23 µg L⁻¹ in 1992 to peaks around 50 µg L⁻¹ in 1996 and 1997 in Meiliang Bay, and then declined to around 20 µg L⁻¹ during 1998–2007 (Fig. 2E). After the drinking water crisis in 2007, *Chla* concentrations continued to increase. Annual average *Chla* concentrations in the central lake were lower than those in Meiliang Bay, ranging from 6.1 µg L⁻¹ in 1998 to 22.9 µg L⁻¹ in 2009 (Fig. 2F). Seasonally, *Chla* concentrations were low (less than 20 µg L⁻¹) during spring and winter, rapidly increased with rising water temperatures, and reached peak values during summer in most years, with a mean of 45 µg L⁻¹. An exception to this pattern occurred in Meiliang Bay during both 1996 and 1997, when spring *Chla* concentrations exceeded 50 µg L⁻¹ and winter *Chla* concentrations reached up to 40 µg L⁻¹ in 1997. Generally, *Chla* concentrations were lower in the central lake than in Meiliang Bay.

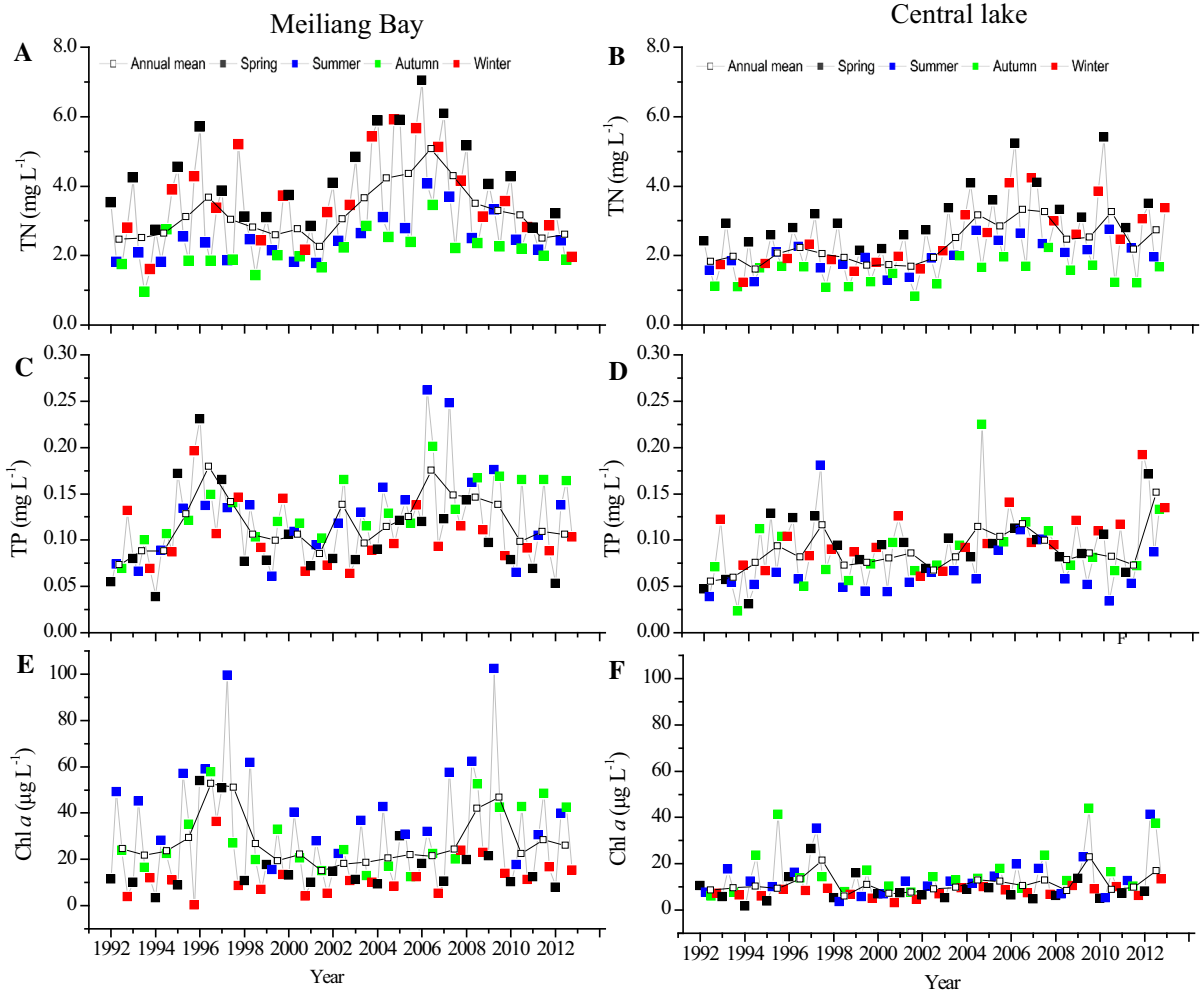


Fig. 2 Long-term trends of annual averaged and seasonal averaged total nitrogen (TN), total phosphorus (TP), and chlorophyll *a* (Chl *a*) concentrations in Meiliang Bay and the central lake region of Taihu

Microcystis bloom biomass steadily increased in magnitude and duration from 1992 to 1998, and remained at high levels up to 2010 (Fig. 3). The *Microcystis* blooms generally began in late spring (April–May), reached a biomass peak in late summer (July–September), and persisted until late fall (November) (Fig. 3A). During the bloom period (late spring to fall), *Microcystis* generally dominated the phytoplankton, accounting for 40–90% of total phytoplankton biomass (Fig. 3C). Generally, *Microcystis* biomass in the central lake was lower than that in Meiliang Bay. However, its contribution to total biomass still remained high.

Relationships of bloom indicators and nutrient concentrations

Long-term development of phytoplankton yields (as Chl *a*) per unit of TN and TP is shown in Fig. 4. Annual average Chl *a* yield per mg TN increased from 1992 to 1996 and then declined from 1997 to 2006 at both locations. Chl *a* yield per unit TN was low in spring and winter, with a mean value of 3 µg Chl *a* per mg TN in Meiliang Bay. However, during spring in 1996 and 1997, Chl *a* yield reached up to about 10 µg per mg TN. Chl *a* yield reached peak values during summer, with a mean value of 19.0 µg Chl *a* per mg TN in Meiliang

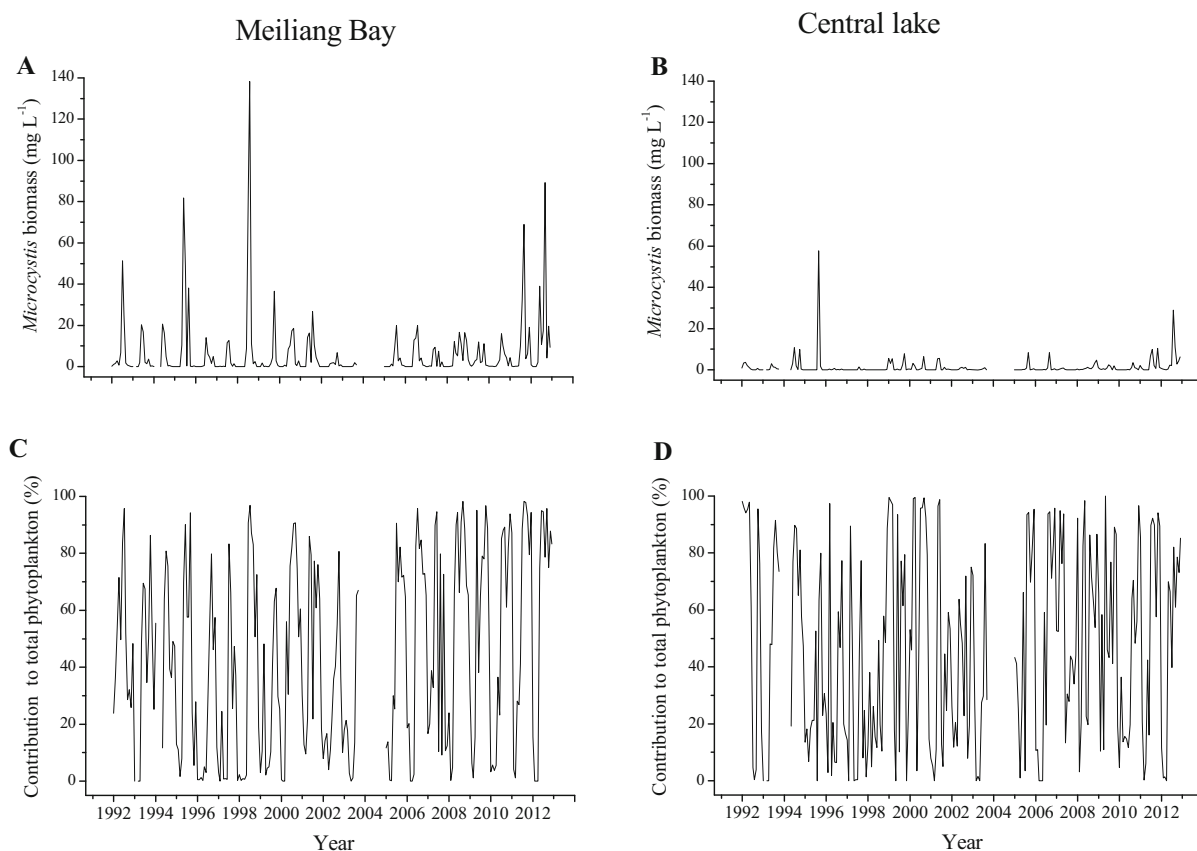


Fig. 3 Temporal trends of monthly averaged (A) *Microcystis* biomass and (B) relative contribution of *Microcystis* biomass to total phytoplankton in Meiliang Bay and the central lake region

Bay. Annual average Chla yield per mg TP declined from 1998 to 2006. Similar to Chla yield per mg TN, Chla yield per mg TP also showed high values during summer in most years, with a mean value of 350 μg Chla per mg TP in Meiliang Bay and 213 μg Chla per mg TP in the central lake. Chla yields per mg TP during spring and winter were low in Meiliang Bay, with mean values of 160 μg Chla per mg TP in spring and 123 μg Chla per mg TP in winter. In the central lake, the mean Chla yields were 100 μg Chla per mg TP in spring and 80 μg Chla per mg TP in winter.

The regressions of annual mean Chla on annual mean TN and TP for the full twenty-one year data record (1992–2012), and for four individual time periods (1992–1996, 1997–2001, 2002–2006, and 2007–2012) are shown in Fig. 5. Annual mean Chla correlated significantly (linear regression) with annual average TP and TN ($P < 0.001$) when analyzed using the dataset covering the entire period (1992–2012). Correlations between annual mean TP

and Chla were stronger than for TN and Chla (Fig. 5). The strength of the relationship between TN and Chla changed over the study period with an initially strong relationship from 1992 to 2001, followed by a weak, but statistically significant, correlation from 2002 to 2006, and finally no significant relationship from 2007 to 2012. TP always showed significant correlations with Chla over the four observational periods ($P < 0.001$) (Fig. 5).

Mean summer Chla showed no relationship with spring TN and TP when analyzed using data from the entire study period (1992–2012) (Fig. 6). During the individual 5-year periods, summer Chla showed a weak positive relationship with spring TN during 1997–2001 ($P < 0.05$) and a stronger positive relationship during 2002–2006 ($P < 0.001$). Summer Chla only showed a weak positive relationship with spring TP during 1997–2001 ($P < 0.01$).

The regressions of summer mean Chla on summer mean TN and TP are plotted in Fig. 7. Summer Chla was

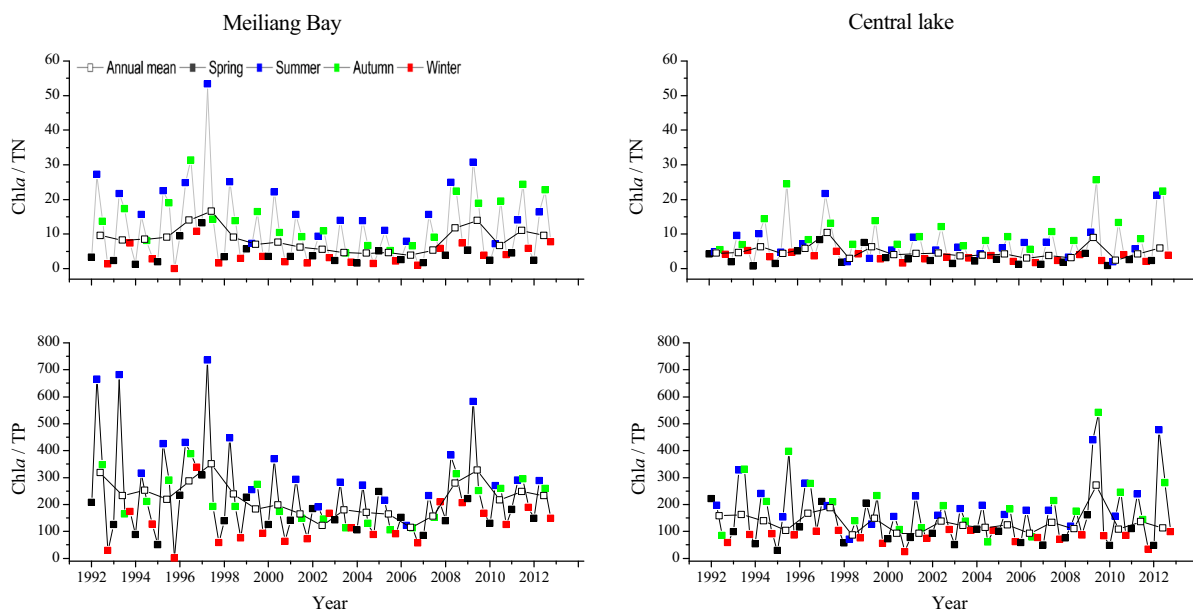


Fig. 4 Temporal trends of annual averaged and seasonal averaged Chl a:TN and Chl a:TP ratio in Meiliang Bay and the central lake region

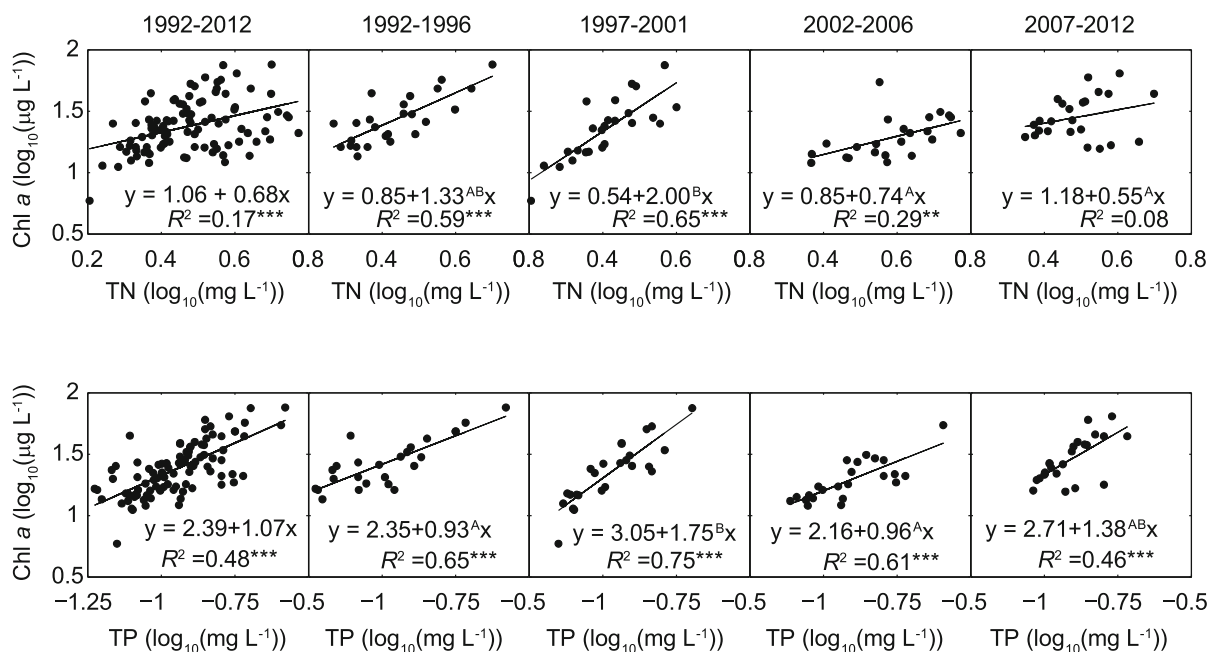


Fig. 5 Linear regressions of \log_{10} annual mean chlorophyll *a* (Chl *a*) on \log_{10} annual mean total nitrogen (TN) and total phosphorus (TP) at stations one through five for the full twenty-one year data record (1992–2012), and for four time periods of roughly equal length. Superscripts *, **, *** following R^2 values indicate P -values for the regression are less than 0.05,

0.01, and 0.001, respectively. Different alphabetic superscripts above regression coefficients indicate statistically significant differences between slopes between the four periods as determined by an analysis of covariance followed by a Tukey–Kramer post hoc test (e.g., A is different than B, AB is similar to A and B)

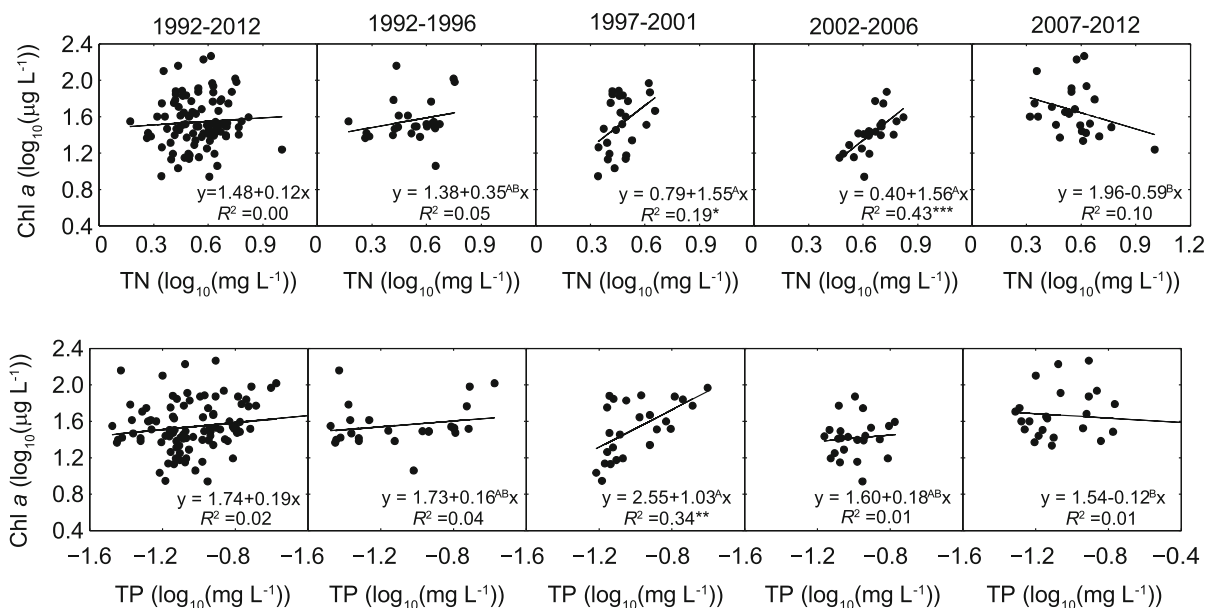


Fig. 6 Linear regressions of log₁₀ summer mean chlorophyll *a* (Chl*a*) on log₁₀ spring mean total nitrogen (TN) and total phosphorus (TP) at stations one through five for the full twenty-one year data record (1992–2012), and for four time periods of roughly equal length. Superscripts *, **, *** following R^2 values indicate P -values for the regression are less than 0.05,

0.01, and 0.001, respectively. Different alphabetic superscripts above regression coefficients indicate statistically significant differences between slopes between the four periods as determined by analysis of covariance followed by a Tukey–Kramer post hoc test (e.g., A is different than B, AB is similar to A and B)

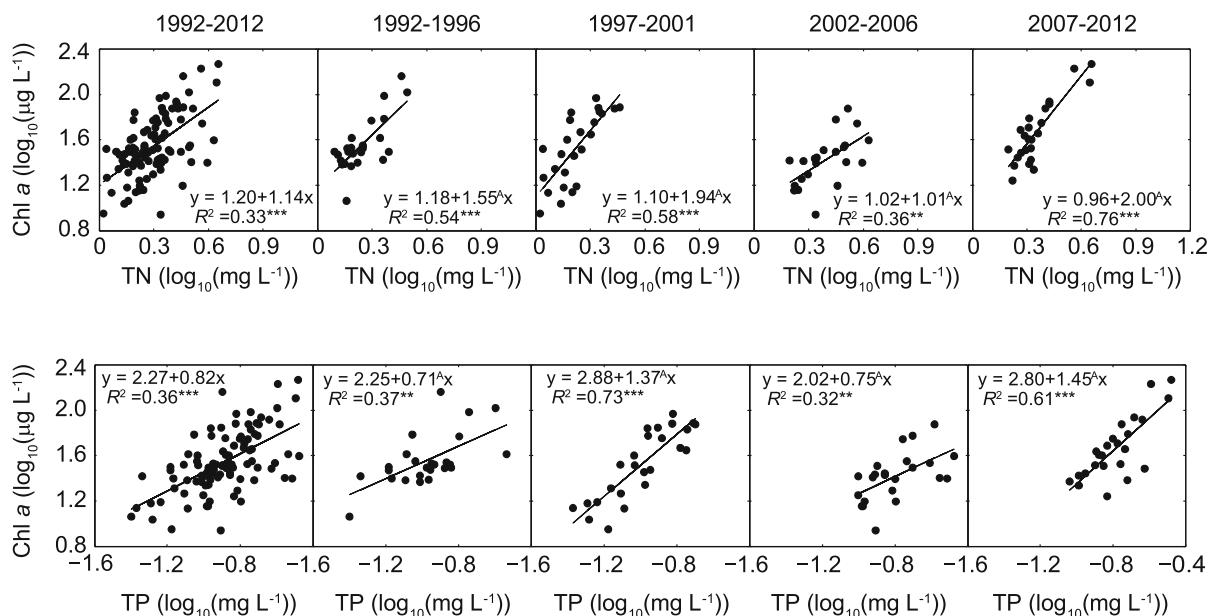


Fig. 7 Linear regressions of log₁₀ summer mean chlorophyll *a* (Chl*a*) on log₁₀ summer mean total nitrogen (TN) and total phosphorus (TP) at stations one through five for the full twenty-one year data record (1992–2012), and for four time periods of roughly equal length. Superscripts *, **, *** following R^2 values indicate P -values for the regression are less than 0.05,

0.01, and 0.001, respectively. Different alphabetic superscripts above regression coefficients indicate statistically significant differences between slopes between the four periods as determined by analysis of covariance followed by a Tukey–Kramer post hoc test (e.g., A is different than B, AB is similar to A and B)

strongly related with summer TN concentrations during 1992–2001 and 2007–2012 ($P < 0.001$), and a weaker significant positive relationship was observed during 2002–2006 ($P < 0.01$). Summer TP showed a significant relationship with summer Chl a during the study period ($P < 0.001$), with especially strong relationships during 1997–2001 and 2002–2012 ($P < 0.001$).

The frequency of Chl a exceeding the bloom threshold of $20 \mu\text{g L}^{-1}$ increased to 50% when TN concentration reached 1.03 (95% CI $0.34\text{--}1.44$) mg L^{-1} (Fig. 8A). There was an increase in Chl a exceedances at TP concentrations ranging from 0.04 to 0.11 mg L^{-1} . At TP concentrations above 0.081 (95% CI $0.072\text{--}0.090$) mg L^{-1} , the frequency of Chl a exceedances reached around 50% (Fig. 8B).

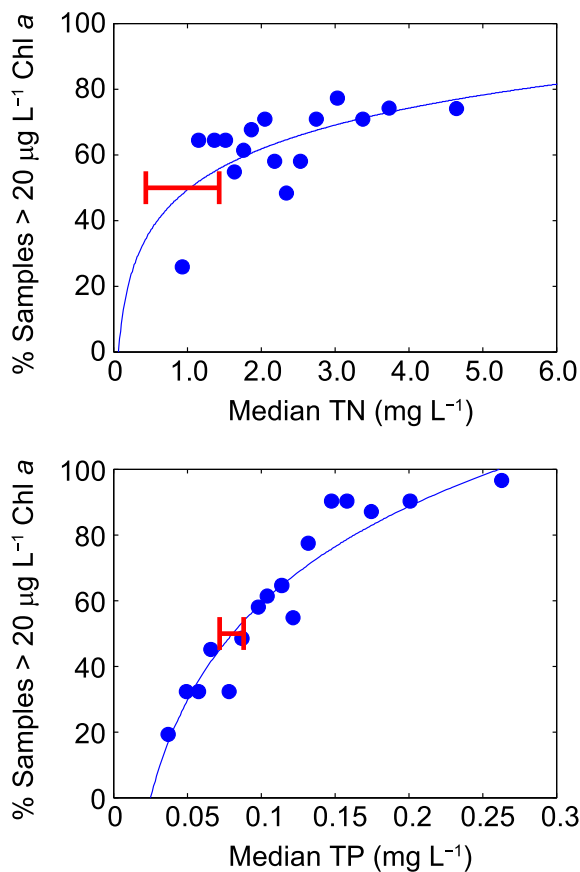


Fig. 8 Relationship between median total nitrogen (TN) (A) and median total phosphorus (TP) (B) within equal sized bins versus the frequency of chlorophyll a (Chl a) exceeding 20 mg L^{-1} . Solid blue line is a best fit to a log-linear model. Red brackets delineate the bootstrapped 95% confidence interval around the TN or TP value related to a chlorophyll a exceedance frequency of fifty percent

Discussion

Anthropogenic influences on water quality of Taihu

Spatial and seasonal distributions of Taihu's TN and TP concentrations corresponded with nutrient loads from the major adjoining tributaries (Li et al., 2011b), suggesting that changes in water quality of the lake mainly depend on external loading. In the 1960s, industrial development was initiated in the Taihu Basin, with TN pollution loads of less than $10,000 \text{ t/a}$ and TP pollution loads less than $1,000 \text{ t/a}$ (Lai & Yu, 2007). During the 1980s–1990s period, industries rapidly expanded in the Taihu Basin and urbanization rate rapidly increased from 35.5% in 1980 to 66.5% in 2000 (Jin et al., 2006). This has led to large increases in industrial and domestic wastewater discharge to the lake. By 1988, TN load had reached $\sim 28,000 \text{ t/a}$ and TP load had reached $\sim 2,000 \text{ t/a}$ (Huang et al., 2001). Taihu was categorized as oligotrophic in 1960, mesotrophic in early 1980s, and eutrophic in late 1980s (Chen et al., 2003b). Furthermore, after 1990, the decline of water quality in Taihu accelerated (Qin et al., 2004). This can be confirmed by the fact that TN and TP concentrations in Taihu rapidly increased from 1992 to 1996.

The decline in TN and TP nutrient concentration from 1997 to 2001 might partly be explained by the reduction of external nutrient input from the catchment imposed by the local government in 1995 (Chen et al., 2003b). The industries in the catchment, especially those located in former rural areas, were forced to control their wastewater discharge (Chen et al., 2003b). In 1998, the local government implemented “zero point action”; this action was aimed at industrial pollution control and included a phosphate detergent ban in the Taihu Basin. These measures have led to a considerable decline in nutrient concentrations after 1996. However, this improving trend in nutrient inputs only lasted until 2001, after which wastewater discharge increased again (Qian & He, 2009). Consequently, nutrient concentrations also increased from 2002 to 2007. Since 2002, water transfer from the Yangtze River was implemented to improve water quality. However, this effort had no notable positive effects on reducing TN and TP concentrations (Hu et al., 2008; Li et al., 2011a). In May 2007, a very large “cyanobacteria mat” caused the Wuxi city drinking

water plant to cease functioning and led to a highly publicized drinking water crisis (Qin et al., 2010). This event prompted the Chinese central and local governments to undertake a wide range of activities in an effort to reduce external nutrient loading including shutting off Liangxi River from Meiliang Bay, closing a number of polluting factories, and building additional wastewater treatment plants (Wu & Hu, 2008; Xin, 2009; Yang & Liu, 2010). The current annual external TN and TP loads are estimated to be around 24,000 and 1,000 t/a, respectively (Li et al., 2011b). These actions improved water quality and have led to declines in TN and TP concentrations after 2007.

Cyanobacterial bloom potential response to nutrient changes in Taihu

Harmful cyanobacterial blooms are a troubling symptom of increasingly eutrophic conditions in lakes (Paerl & Otten, 2013). It is important to examine cyanobacterial bloom potential response to long-term nutrients change in Taihu. Correlation analyses showed that annual mean TP was always significantly positively related with *Chla* using the data set covering the entire 1992–2012 period or from each of the four individual periods ($P < 0.001$). This suggests that phytoplankton biomass had a strong response to annual changes in TP, and that TP was a good predictor of *Chla* on an annual scale. Annual mean TN showed strong significant positive relationships with *Chla* during both periods of 1992–1996 and 1997–2001 ($P < 0.001$). This indicates that the increase and subsequent decrease of external N loading had a significant effect on phytoplankton biomass during these two periods. However, annual mean TN showed a slightly weaker but still significant relationship with *Chla* during 2002–2006 ($P < 0.01$), and no relationship was observed during 2007–2012, indicating that phytoplankton biomass responded slowly to annual changes in TN pollution during these periods. Chlorophyll *a* generally showed higher values during summer and fall, and lower values during winter and spring. However, TN concentrations were higher during winter and spring, and declined to lower values in summer and autumn during last 20 years. Therefore, higher N concentrations did not coincide with higher *Chla* concentrations in most periods. This probably strongly affected the relationships between annual mean *Chla* and annual mean TN.

Summer *Chla* showed weak relationships with spring TN, indicating that a large fraction of the TN loss could not be accounted for by increases in *Chla*. Denitrification is a potentially important N loss mechanism that can affect nutrient supply ratios and drive aquatic systems toward N limitation (Seitzinger, 1988; Seitzinger et al., 2002; McCarthy et al., 2007). This sediment-based process is especially important in a shallow lake like Taihu due to the lake's very high sediment surface area to volume ratio. It has been estimated that on an annual basis, ~60% of external N loading is "lost" by denitrification in Taihu (Paerl et al., 2011). The majority of N loss due to denitrification occurs during spring when nitrate is abundant and temperatures begin to increase (Zhang L., personal communication). Simple calculations based on summer *Chla* yield and the spring time TN and DIN pools provide estimates of the magnitude of reduced summertime bloom potential that is likely due to removal of N by denitrification. The current mean spring TN concentration was 3.7 mg L^{-1} and summertime *Chla* yield on TN is $\sim 19 \text{ } \mu\text{g L}^{-1} \text{ Chla per mg TN}$. If we assume no significant N losses due to denitrification, uptake of the spring time N pool would be expected to allow for the development of $70 \text{ } \mu\text{g L}^{-1} \text{ Chla}$ during the summertime bloom. If the expected yield is calculated using spring mean DIN of 2.6 mg L^{-1} (data not showed) and the relationship between summer phytoplankton biomass and DIN concentration ($\text{Chla} = 24.38\text{DIN} + 0.43$) (Xu et al., 2015), the spring time DIN pool would be expected to support development of $68 \text{ } \mu\text{g L}^{-1} \text{ Chla}$. Both results are very similar. The fact that mean summer *Chla* concentration is only $45 \text{ } \mu\text{g L}^{-1}$, suggests that about 35% of the spring time N inputs that could have fueled biomass development was instead lost due to denitrification. Losses due to denitrification in the sediments likely help to explain the relatively weak relationships observed between spring TN and summertime *Chla*. Similarly, sediment processes likely weaken the relationship between spring TP and summer *Chla*. Sediments retain P from winter to spring, but release it during summer (Köhler & Gelbrecht, 1998). Consequently, lower TP concentrations during winter and spring and higher TP concentrations during summer were observed in Taihu in most years. This strongly affect the relationship of summer *Chla* with spring TP. Seasonal in situ nutrient addition bioassays indicate that algal production was limited by P

availability during spring (Xu et al., 2010). However, there is an exception during 1996–1997, the concentrations of SRP were high during spring in both years, leading to an unusual spring bloom of chlorophytes with $\sim 50 \mu\text{g L}^{-1}$ Chla (Chen et al., 2003a). Therefore, controlling spring P input will help to constrain spring algal bloom potential.

Summer Chla showed strong relationships with summer TN and TP during 1992–2012, indicating bioavailability of both N and P during the summer season plays an important role for sustaining the cyanobacterial bloom. Nutrient addition bioassays also confirmed that N and P availability control production during the summer-fall when Cyanobacteria are most severe (Xu et al., 2010; Paerl et al., 2011). Therefore, a dual nutrient (N and P) reduction strategy is needed to control eutrophication and bloom outbreaks in Taihu (Paerl et al., 2011). The seasonal variability of nutrient loads to Taihu showed maximum values in summer (Li et al., 2011b). Consequently, reducing TN and TP inputs is an essential step for controlling summer blooms.

Nutrient thresholds for controlling cyanobacterial blooms in Taihu

Cyanobacterial blooms have been observed in some regions of Taihu since the 1960s, but the frequencies, intensities, and aerial extent of events were relatively low during that period; hence, water quality and resources were not significantly affected (Qin et al., 2004). In the early 1970s, cyanobacterial blooms appeared in Wuli Bay near Wuxi, after which their magnitude and frequencies have increased. In the mid- to late-1980s, algal blooms occurred two to three times yearly, and expanded into Meiliang Bay. In the middle and late 1990s, algal blooms occurred four to five times yearly and gradually expanded to most of the rest of the lake. By 2000, the central region of Taihu had become more impacted by cyanobacterial blooms. After the drinking water crisis in 2007, measures to control eutrophication have decreased the nutrient levels, especially for N, in the lake. However, the severity of blooms in Taihu has not lessened, and cyanobacterial blooms extending to areas of hundreds of km^2 remain common in the summer (Qin et al., 2015).

Clearly, there is a need for developing nutrient-bloom thresholds to help mitigate the degrading

ecological condition, by controlling inputs and availability of both N and P. Chlorophyll *a* is a commonly used proxy for algal biomass and has been proposed as a criterion for identifying water bodies that fail to attain their designated use(s) due to excessive nutrient loading. From a human recreational and fisheries use perspective, impairment is more closely linked to the frequency and intensity of algal blooms than annual mean Chla concentration. Chlorophyll *a* concentrations exceeding $20 \mu\text{g L}^{-1}$ are commonly observed during cyanobacterial bloom period (Xu et al., 2015); therefore, we can use Chla concentrations exceeding $20 \mu\text{g L}^{-1}$ as the criteria for bloom conditions.

Chlorophyll *a* yield during summer showed a mean value of $19.0 \mu\text{g Chla per mg TN}$ and $350 \mu\text{g Chla per mg TP}$ in Meiliang Bay (Fig. 5). Using $20 \mu\text{g L}^{-1}$ as the base condition for bloom formation, required TN and TP concentrations are 1.05 and 0.057 mg L^{-1} , respectively. When TN and TP concentrations reach 1.03 and 0.08 mg L^{-1} , respectively, the frequency of Chla exceeding $20 \mu\text{g L}^{-1}$ reached around 50%. These thresholds are similar to results from Chla yield. Therefore, 1.0 mg L^{-1} TN and 0.08 mg L^{-1} TP can be considered as thresholds for controlling the frequency and intensity of harmful cyanobacterial blooms. The water quality targets set by the Chinese Central Government for Taihu are TN concentration below 1.20 mg L^{-1} and TP concentration below 0.050 mg L^{-1} by 2020 (Cheng et al., 2013). These water quality targets will ensure “safe water quality” in line with the nutrient thresholds developed here. When examining the history of nutrient increases in Taihu, by 1981, TIN had increased to 0.89 mg L^{-1} and SRP was 0.02 mg L^{-1} . In 1989, TN increased to 2.48 mg L^{-1} and TP increased to 0.07 mg L^{-1} (Sun & Huang, 1993). Our currently developed threshold concentrations are close to those observed in late 1980s. We therefore conclude that, based on these nutrient thresholds, achieving significant reduction in bloom potential and long-term bloom control in Taihu will require reducing TN and TP levels to those observed in mid- to late-1980s.

Conclusion

Changes in water quality of Lake Taihu largely depend on changes in external loading. The TN and TP concentrations during period of 1992–2012 showed four periods of change in response to anthropogenic

influences. Annual average TN and TP concentrations first increased from 1992 to 1996 and then declined from 1997 to 2001. The concentrations increased again from 2002 to 2006 and then declined to the 1992 level. Phytoplankton biomass showed strong responses to annual change of TP, and TP provided a good predictor of *Chla* on the annual scale. However, phytoplankton biomass responded slowly to annual change of N pollution after 2002. Spring TN showed a weak relationship with summertime *Chla*. The loss of a significant fraction of spring N loading due to denitrification likely reduces summertime bloom biomass development. The bioavailability of both N and P during the summer season plays an important role in sustaining the cyanobacterial bloom. TN and TP thresholds can be set as 1.0 and 0.08 mg L⁻¹ for controlling the frequency and intensity of CyanoHABs in Taihu.

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