Knowl. Manag. Aquat. Ecosyst. 2022, 423, 11
© L. Zhang *et al.*, Published by EDP Sciences 2022
https://doi.org/10.1051/kmae/2022009

www.kmae-journal.org

Knowledge &
Management of
Aquatic
Ecosystems

Journal fully supported by Office français de la biodiversité

RESEARCH PAPER OPEN ACCESS

Effects of Nile Tilapia (*Oreochromis niloticus*) on phytoplankton community structure and water quality: a short-term mesocosm study

Liqiong Zhang[1,2,a], Xueying Mei[3,a], Yali Tang[1,2], Vladimir Razlutskij[4], Jiří Peterka[5], William D. Taylor[6], Luigi Naselli-Flores[7], Zhengwen Liu[1,8,9], Chunfu Tong[10,*] and Xiufeng Zhang[1,2,*]

[1] Department of Ecology and Institute of Hydrobiology, Jinan University, Guangzhou 510632, China
[2] Engineering Research Center of Tropical and Subtropical Aquatic Ecological Engineering, Ministry of Education, Guangzhou, China
[3] College of Resources and Environment, Anhui Agricultural University, Hefei 230036, China
[4] State Scientific and Production Amalgamation Scientific-Practical Center of the National Academy of Sciences of Belarus for Biological Resources, Minsk, Belarus
[5] Department of Fish and Zooplankton Ecology of the Institute of Hydrobiology, Biology Centre of the Czech Academy of Sciences of the Czech Republic, České Budějovice, Czech Republic
[6] Department of Biology, University of Waterloo, Waterloo, ON N2L 3G1, Canada
[7] Department STEBICEF, University of Palermo, 90133 Palermo, Italy
[8] Sino-Danish Centre for Education and Research (SDC), Beijing 100070, China
[9] State Key Laboratory of Lake Science and Environment, Institute of Geography and Limnology, Chinese Academy of Sciences, Nanjing 210008, China
[10] State Key Laboratory of Estuarine and Coastal Research, East China Normal University, Shanghai 200062, China

Received: 14 November 2021 / Accepted: 12 March 2022

Abstract – Nile tilapia is a highly invasive fish species, deliberately introduced into many lakes and reservoirs worldwide, sometimes resulting in significant ecosystem alterations. A short-term mesocosm experiment with and without Nile tilapia (*Oreochromis niloticus*) was designed to test the hypotheses that the presence of tilapia may affect phytoplankton community structure, increase nutrients availability in water column and deteriorate water quality. Nutrients, total suspended solids (TSS) and biomass of phytoplankton in different size classes (as Chl *a*) were measured. We found that tilapia increased the total nitrogen (TN), total dissolved nitrogen (TDN), NH_4^+ and TSS concentrations, deteriorating the water quality. In addition, under tilapia presence, the biomass of phytoplankton, as well as that of micro- and nano-phytoplankton, increased leading to a change in the structure of the phytoplankton assemblage. Moreover, a reduction in the biomass of periphyton was observed. Omnivorous tilapia is often dominant in tropical and subtropical waters, and removal of this fish may represent an effective management tool to improve the water quality.

Keywords: Tilapia / nutrient / phytoplankton community / periphytic algae / water quality

1 Introduction

The Nile Tilapia (*Oreochromis niloticus*), an omnivorous fish species, is native to Africa and the southwestern Middle East. It feeds on a variety of foods, including, among others, phytoplankton, zooplankton, suspended detritus, macrophytes and insect larvae (Gurgel and Fernando, 1994; Drenner *et al.*, 1998; Starling *et al.*, 2002; Canonico *et al.*, 2005). Due to its wide tolerance to environmental constraints, plasticity in feeding behavior, high reproductive rate, year-round spawning and parental brood care, tilapia is considered an extremely invasive species (Kolding, 1993; Canonico *et al.*, 2005; Kour *et al.*, 2014). For nearly a century, it has been introduced in many lakes, reservoirs and rivers worldwide for biological control of insects (especially mosquitos) and aquatic weeds, as bait for some fisheries, and for aquaculture purposes (Starling *et al.*, 2002; Canonico *et al.*, 2005; Figueredo *et al.*, 2005).

Tilapia invasion into non-native freshwater ecosystems may alter phytoplankton community structure, nutrient

[a] These authors contributed equally to this work.
*Corresponding author: cftong@sklec.ecnu.edu.cn; wetlandxfz@163.com

availability and water quality, causing the deterioration of the ecological state of the impacted ecosystems (Peterson *et al.*, 2005; Vicente and Fonseca-Alves, 2013; Charvet *et al.*, 2021). Some data show that omnivorous tilapia can reduce phytoplankton biomass by direct consumption (Saha and Jana, 1998; Lazzaro *et al.*, 2003; Figueredo and Giani, 2005; Lu *et al.*, 2006). The fish can feed on large phytoplankton (Getachew and Fernado, 1989; Turker *et al.*, 2003a; Turker *et al.*, 2003b) using a remarkable array of specialized gill rakers and a cross-flow filtration system (Sanderson *et al.*, 2001). However, some other studies have shown that the omnivorous tilapia may enhance phytoplankton growth in tropical lakes (Attayde and Menezes, 2008).

Omnivorous tilapia, particularly juveniles (Tudorancea *et al.*, 1988), can also consume zooplankton, which is the principal grazer of phytoplankton (Hansson, 1992a; Jeppesen *et al.*, 2004; Figueredo and Giani, 2005), thus exerting an effective control of phytoplankton abundance (Cordero *et al.*, 2015). In fact, when large-bodied zooplankton, such as *Daphnia*, is abundant, phytoplankton biomass is generally low (Lynch and Shapiro, 1981; Attayde and Hansson, 1999; Vakkilainen *et al.*, 2004). Tilapia feeding selectively on zooplankton prefers large-bodied zooplankton including cladocerans and copepods, shifting zooplankton community towards smaller species which are relatively inefficient grazers (Christoffersen *et al.*, 1993; Iglesias *et al.*, 2011; Chen and Liu, 2012). Thus, this consumption of tilapia may result in changes in zooplankton community structure which not only affects the biomass of phytoplankton, but also its community structure due to the selective feeding by zooplankton of different size.

Tilapia is also able to affect nutrient concentration in water column through its excretions and sediment disturbance through snuffling. Nutrients availability affects phytoplankton growth and ultimately the structure of phytoplankton assemblages (Persson, 1997; Vanni and Layne, 1997; Marañón, 2015). In particular, increased nutrient availability, by enhancing phytoplankton growth, causes a dominance shift from small cells to large cells (Li, 2002; Chen and Liu, 2010, Cui *et al.*, 2016).

Furthermore, omnivorous fish including tilapia can also consume periphyton which can compete for nutrients with phytoplankton, thereby favoring phytoplankton growth (Guariento *et al.*, 2010), but may potentially also improve macrophyte growth due to improved light conditions (Meerhoff *et al.*, 2012).

For the above listed reasons, tilapia may not only affect the phytoplankton biomass, but also the structure of its assemblages, and, ultimately, the water quality. Phytoplankton dynamics is a key factor affecting food web, water quality and even the ecosystem state (Kormas *et al.*, 2002; Finkel *et al.*, 2010; Hilligsøe *et al.*, 2011; Shiomoto *et al.*, 2018). However, the effects of tilapia invasion on the structure of plankton community and water quality are still controversial, and deserve further studies. Investigating the biological interactions arising from the presence of tilapia on the plankton community structure can offer important clues to better understand the functioning of aquatic ecosystems impacted by the invasion of this fish. We hypothesize that tilapia can (i) significantly increase nutrient concentration in the water column, (ii) cause a general worsening of water quality, and (iii) strongly affect plankton community structure by consuming large phyto- and zooplankton. To test the hypotheses, mesocosms with and without tilapia were set up, to analyze the effect of fish presence on water quality parameters and on Chl *a* concentrations of different phytoplankton size-fractions.

2 Materials and methods

2.1 Set up of experimental mesocosms

Eight circular tanks (upper diameter, 57 cm; bottom diameter, 46 cm; height, 82 cm) were used as mesocosms containing sediment and water. The sediments ($TN = 1.13\ mg\ g^{-1}$; $TP = 0.56\ mg\ g^{-1}$) were collected from Ming Lake (23°13′70″-23°13′75″ N, 113° 35′62″-113° 35′74″ E), a small shallow eutrophic lake in Guangzhou, China, air dried, powdered and screened through a 0.5 mm stainless steel mesh to remove coarse grains, debris and clumps (Zhang *et al.*, 2016). The water consisted of collected rainwater ($TN = 1.58\ mg\ L^{-1}$; $TP = 0.008\ mg\ L^{-1}$). Each mesocosm had a 10 cm layer sediment added, then was filled with the collected rainwater and left untouched for two weeks.

After this period, in all the mesocosms the concentration of TN in the water column was similar to that of the rainwater, while TP increased to $0.011\ mg\ L^{-1}$. An artificial leaf made of plastic (length 22.0 cm, width 1.2 cm) was placed on the sediments of each mesocosm as a substate for periphyton growth. Then, one individual Nile tilapia *Oreochromis niloticus* (average length 12.6 ± 0.2 cm and weight 53.6 ± 4.2 g) was added in each of four mesocosms that represented the tilapia treatments. The fish, purchased from a market in Guangzhou, were maintained in 200-L tanks for two weeks before being introduced in the mesocosms. The tilapias were not fed and the mesocosms were checked every day. In the event of dead fish, they were replaced with a new specimen immediately. Two individuals died at the fourth day of the experiment and were replaced with living specimens of the same size. The other four mesocosms contained no fish as controls. Nitrogen (KNO_3) and phosphorus (NaH_2PO_4) were added to each mesocosm at a rate of $1.5\ mg\ N\ L^{-1}\ wk^{-1}$ and $0.1\ mg\ P\ L^{-1}\ wk^{-1}$, respectively (Zhang *et al.*, 2014). During the experiment, rainwater was added when needed to maintain the water level in each mesocosm. The experiment was carried outdoor from August 18 to October 27, 2020.

2.2 Sampling and analysis

Every two weeks, water samples (1L) were collected from 10 cm below the water surface of each mesocosm to measure nitrogen (TN, TDN, NO_3^--N, NH_4^+-N), different size-classes of phytoplankton biomass and periphyton biomass as Chl *a*, total suspended solids (TSS). Nutrients were measured according to American Public Health Association (APHA, 1998). Chl *a* was measured according to Jespersen and Christoffersen (1987). Water samples (200 mL) were filtered on cellulose acetate filters (0.45 µm, 0.2 µm, 2 µm, 20 µm) to measure the Chl *a* concentration in phytoplankton, pico-phytoplankton, nano-phytoplankton and micro-phytoplankton, respectively (Rong et al., 2017). TSS was measured by drying the filters at 105 °C for 24 h to calculate TSS (Zhang *et al.*, 2017). Water temperature was measured by a YSI Model multi-parameter probe. Periphyton growing on the artificial leaf from each mesocosm was collected using a soft brush. After samples of water and periphyton were collected, nutrients were added to each mesocosm and the artificial substrates for periphyton growth were replaced. The weight and length of each fish were recorded at the end of the experiment.

2.3 Statistical analyses

Repeated measures analysis of variance (RM-ANOVAs) was used to assess the differences in nutrient concentrations, different sizes of phytoplankton, periphytic algae biomass and TSS between the controls and the tilapia treatments, with time as the repeated factor after a check of the descriptive statistics and homogeneity of variance. One-way analysis of variance (one-way ANOVA) was conducted to analyze the differences on every sampling occasion between treatments. If the difference was significant, Least Significant Difference (LSD) test was used to detect different treatments. Data were $\log_{10}$ transformed, if necessary, to meet the assumptions of normality and homogeneity of variance. Statistical analyses were carried out using IBM SPSS Statistics 26. All data are presented as mean values ± SD (standard deviation).

3 Results

3.1 Nitrogen

Concentrations of TN, TDN and NH_4^+-N, but not NO_3^--N (Fig.1) were higher in the tilapia treatments than in the controls (RM-ANOVAs, treatment effect, $p < 0.05$). In addition, TN, TDN and NO_3^--N varied significantly over time (RM-ANOVAs, time effect, $p < 0.05$), while NH_4^+-N did not. ($p > 0.05$).

3.2 TSS

Concentrations of TSS were higher in the tilapia treatments than in the controls (RM-ANOVAs, treatment effect, $p < 0.05$, Fig. 2), especially at week 2 and week 4 (one-way ANOVA, treatment effect, $p < 0.05$). The TSS concentrations did not vary significantly over time (RM-ANOVAs, time effect, $p > 0.05$).

3.3 Biomass of phytoplankton

Concentrations of Chl *a* in the water column were higher in the tilapia treatments than in the controls (RM-ANOVAs, treatment effect, $p < 0.05$; Fig. 3). Its values were higher in the tilapia treatments than in the controls on each sampling occasion except at week 4 and week 8 (one-way ANOVA, treatment effect, $p < 0.05$). The phytoplankton biomass varied significantly over time (RM-ANOVAs, time effect, $p < 0.05$).

3.4 Biomass of phytoplankton in different size classes

The Chl *a* concentrations in micro-phytoplankton and in nano-phytoplankton were higher in the tilapia treatments than in the controls (RM-ANOVAs, treatment effect, $p < 0.05$, Fig. 4). However, pico-phytoplankton biomass (as Chl *a*) values were generally not significantly different between the treatments (RM-ANOVAs, treatment effect, $p > 0.05$); significantly higher values were recorded in tilapia treatments only at week 6 and week 10 (one-way ANOVA, treatment effect, $p < 0.05$). Micro-phytoplankton biomass varied significantly over time (RM-ANOVAs, time effect, $p < 0.05$), being higher in the tilapia treatments on each sampling occasion (one-way ANOVA, treatment effect, $p < 0.05$), while nano-phytoplankton and pico-phytoplankton biomass did not vary significantly over time ($p > 0.05$).

3.5 Periphytic algae

The chl *a* values of periphytic algae in the tilapia treatments were lower than in the controls (RM-ANOVAs, treatment effect, $p < 0.05$, Fig. 5), especially at week 6 and week 8 (one-way ANOVA, treatment effect, $p < 0.05$). In addition, the chl *a* values varied significantly over time (RM-ANOVAs, time effect, $p < 0.05$).

4 Discussion

The results of our study show that the presence of tilapia increased the concentrations of TN, TDN and NH_4^+, TSS and the biomass of phytoplankton, deteriorating water quality. In addition, in the tilapia treatments, higher values of micro-phytoplankton and nano-phytoplankton biomass occurred, showing a modification in the size-structure of phytoplankton assemblage compared to controls. Moreover, the biomass of periphyton in the treatments with tilapia was lower than in the controls.

As an omnivorous species, tilapia can increase nutrient concentrations in several different ways (Huang *et al.*, 2000; He *et al.*, 2018; Yu *et al.*, 2020). The fish can promote sediment resuspension, and therefore transfer nutrients from the benthic to pelagic habitat, when searching for benthic food or when digging holes for building nest (Jiménez-Montealegre *et al.*, 2002; Vanni, 2002; Mayer, 2020). It can also excrete nutrients directly to the water column (Starling *et al.*, 2002). The increase of TN, TDN and NH_4^+ concentrations observed in the tilapia treatments during our study, strongly support the hypothesis that this fish exerts a positive effect on nutrient availability. Unfortunately, P concentrations were not measured in this study, but previous investigations showed that TP also increases when tilapia is present (Zhang *et al.*, 2017).

It is well known that increased nutrient can enhance growth of phytoplankton (Schindler, 1974) and influence the size-structure of the assemblage (Naselli-Flores, 2014). Phytoplankton assemblages are usually dominated by pico-phytoplankton in nutrient poor conditions (Stockner, 1988; Agawin *et al.*, 2000), while micro-phytoplankton generally dominates in nutrient-rich waters (Stockner *et al.*, 1986; Cui *et al.*, 2016). Tilapia can ingest phytoplankton, especially larger species, such as green algae, diatoms, and even small cyanobacteria (Turker *et al.*, 2003a; Lu *et al.*, 2006). It has been shown that tilapia feeding can significantly reduce phytoplankton number with larger species being filtered proportionally more than smaller ones (Turker *et al.*, 2003b). Furthermore, tilapia can also consume large-bodied zooplankton (Carpenter and Kitchell, 1993; Menezes *et al.*, 2010;

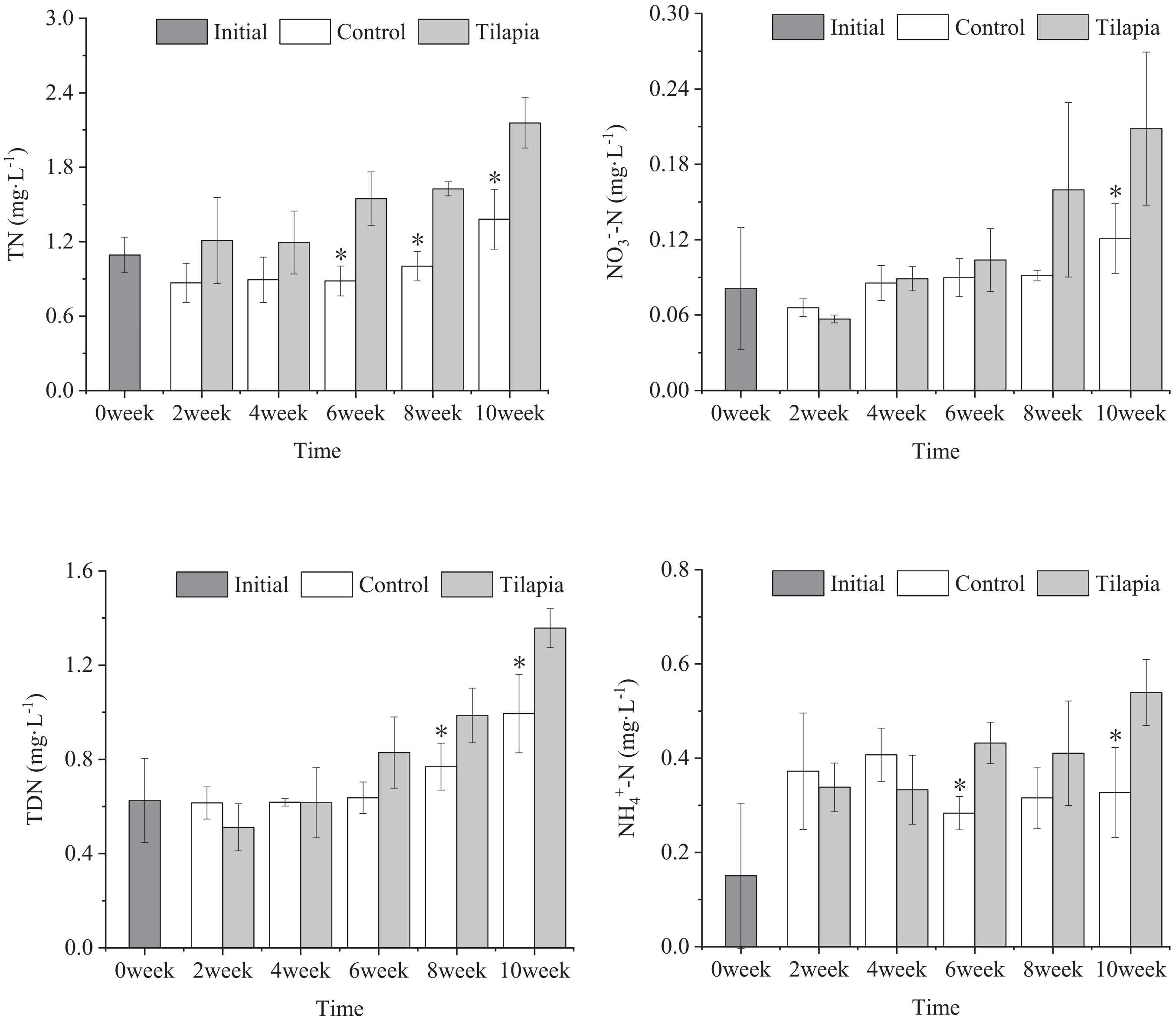

Fig. 1. Nitrogen (TN, TDN, NH_4^+-N, NO_3^--N, mean±SD) in different treatments over time. Asterisks indicate significant ($p < 0.05$) differences between the tilapia treatments and the controls.

Torres *et al.*, 2015) whose grazing is important in controlling the size spectrum of phytoplankton assemblages (Cordero *et al.*, 2015), shifting zooplankton community towards smaller species. Because the body size of an individual zooplankton is well related to its grazing rate and to the range of particle sizes it can ingest (Cyr and Curtis, 1999). So, large-bodied zooplankton, such as *Daphnia*, can exert an effective control on phytoplankton with big size. Conversely, small-sized zooplankton, consuming small phytoplankton, is less able to exert an effective control on phytoplankton (Tõnno *et al.*, 2016). Thus, the shifting of zooplankton community may not only affect the biomass of phytoplankton, but also affect the phytoplankton in different size-classes due to their selected feeding.

Our study shows that the biomass of micro-phytoplankton and nano-phytoplankton in the tilapia treatments increased significantly as compared with the controls, while the biomass of pico-phytoplankton was similar in both the experimental groups of mesocosms. The average proportion of micro-phytoplankton biomass in the control and in the tilapia treatment was 31.3% and 62.5% respectively, while the proportion of pico-phytoplankton in the control and in the tilapia treatments was 59.8% and 29.1%, respectively. However, the proportion of nano-phytoplankton biomasses was relatively stable in both tilapia treatments and controls. The increased proportion of micro-phytoplankton biomass and decreased proportion of pico-phytoplankton biomass in the fish treatments indicated that the fish contributed to modify the size structure of phytoplankton assemblages.

The effects of omnivorous tilapia on biomass of phytoplankton can be either positive or negative (Attayde *et al.*, 2010) depending on trophic state. Some studies have

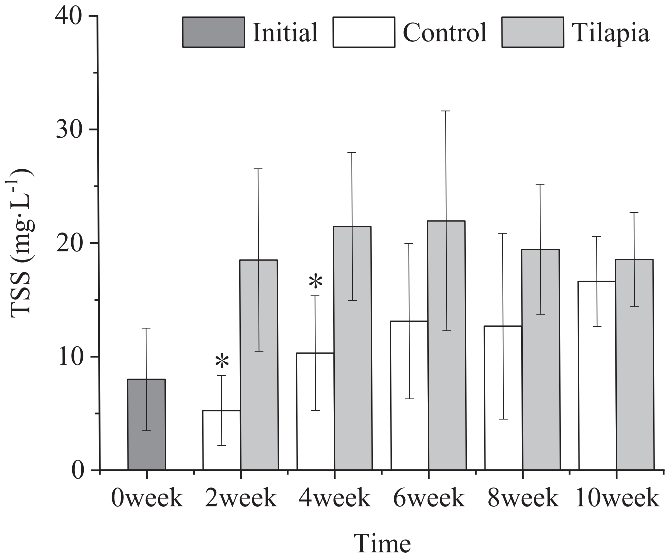

Fig. 2. Total suspended solids (TSS, mean ± SD) in different treatments over time. Asterisks indicate significant ($p < 0.05$) differences between the tilapia treatments and the controls.

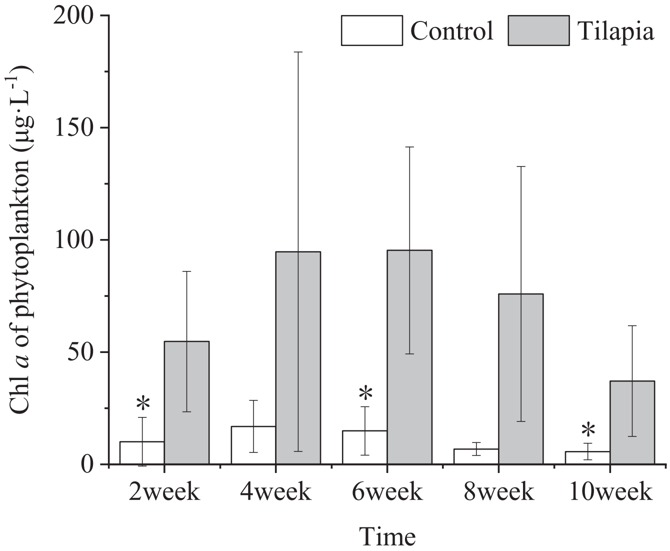

Fig. 3. Phytoplankton biomass (chl *a*, mean ± SD) in the different treatments over time. Asterisks indicate significant ($p < 0.05$) differences between the tilapia treatments and the controls.

shown that tilapia can reduce the biomass of phytoplankton by direct grazing, and tilapia stocking is considered an effective method to control algal blooms in eutrophic waters (Lu *et al.*, 2006; Menezes *et al.*, 2010). Conversely, other studies have confirmed that tilapia negatively affect on water quality, enhancing nutrient availability and phytoplankton growth (Starling *et al.*, 2002; Figueredo and Giani, 2005; Søndergaard *et al.*, 2008; Zhang *et al.*, 2017). The results of the present investigation showed that the presence of tilapia significantly contributed to promote phytoplankton growth.

Our study also showed that tilapia increased the concentration of TSS and decreased periphyton biomass. The increased concentration of TSS also contributes to deteriorating water

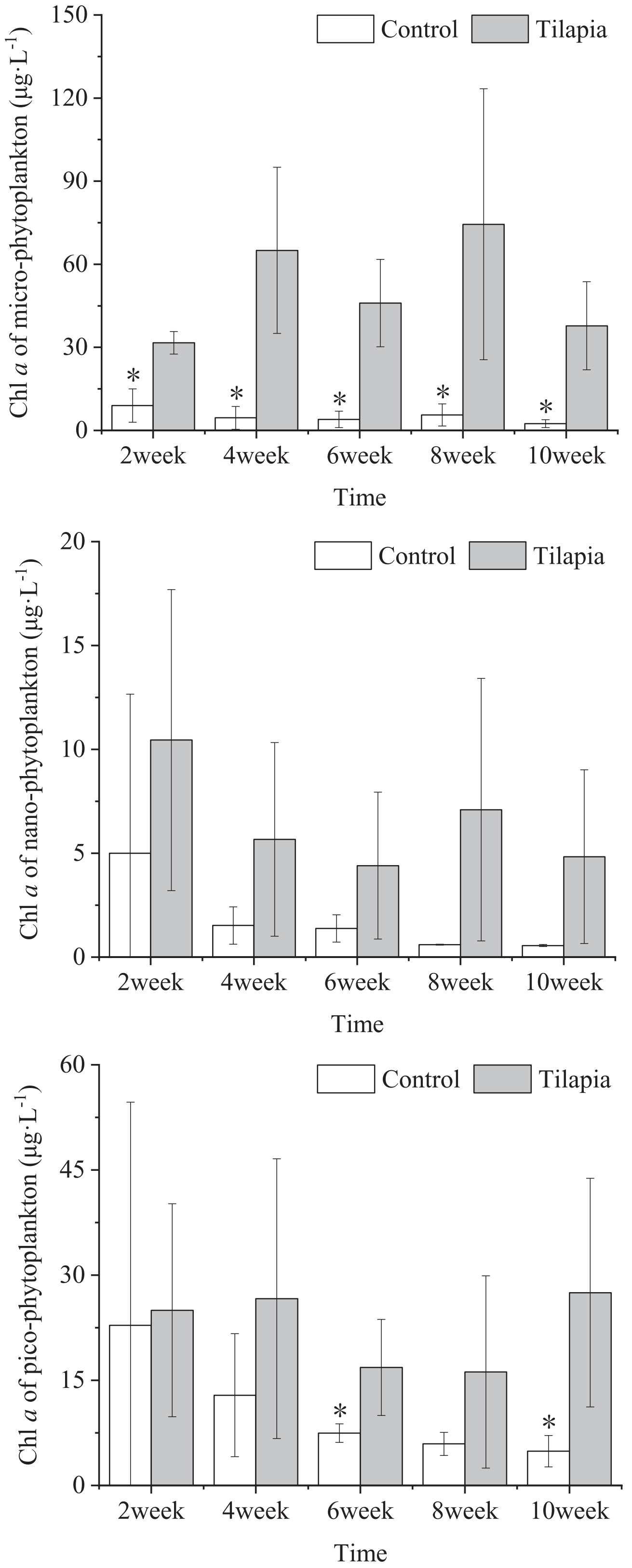

Fig. 4. Micro-phytoplankton, nano-phytoplankton and pico-phytoplankton biomass as chl *a* concentration (mean ± SD) in the different treatments over time. Asterisks indicate significant ($p < 0.05$) differences between the tilapia treatments and the controls.

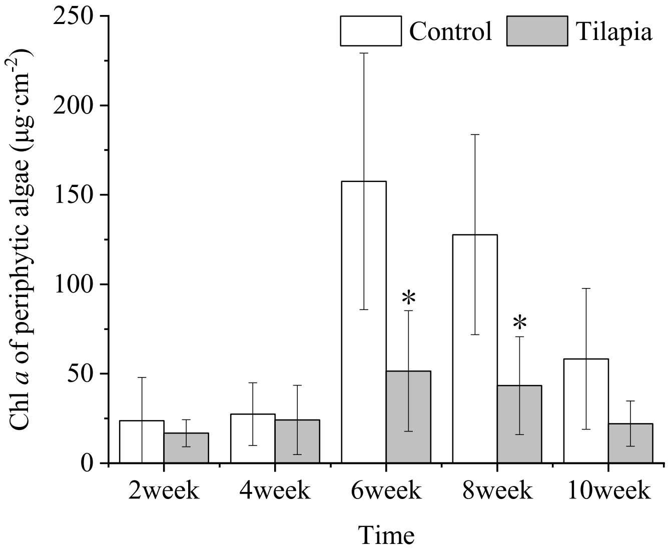

Fig. 5. Chl *a* of periphytic algae (mean ± SD) in the different treatments over time. Asterisks indicate significant ($p < 0.05$) differences between the tilapia treatments and the controls.

quality (Hansson, 1992b; Vadeboncoeur and Carpenter, 2001; Zhang *et al*., 2015; Razlutskij *et al*., 2021). The decreased abundance of periphytic algae may further benefit phytoplankton growth, lowering the competition for sediment-associated nutrients (Hansson, 1990).

No temperature difference was found between treatments and water temperature was 27.8 ± 2.1 °C in the tilapia treatments and 27.4 ± 2.2 °C in the controls. Density of tilapia (322.5 ± 25.2 g m^{-2}) used in this experiments is realistic, as biomass of the fish in aquatic ecosystems can reach to 390–810 g m^{-2} (Suresh and Lin, 1992). Length of the fish increased from 12.6 ± 0.2 cm to 14.9 ± 0.7 cm and weight increased from 53.6 ± 4.2 g to 55.8 ± 10. 6 g during the experiment.

We acknowledge that the results obtained in this study is from a short-term (from August to October) mesocosm experiment. In other seasons, the degrees of the fish impact might vary because of the temperature, light and biological activity. We also acknowledge that applying results from the experimental conditions to natural lakes can be problematic due to the small size of the mesocosm used in this study but mesocosm experiments can be replicated and play an important role in the pursuit of ecological understanding. In addition, the general trend of the influence might be similar. However, tilapia would coexist with other fish species in the real world, these combined effects would influence nutrients and algal growth in natural aquatic ecosystems. So, more research is needed on these combined effects due to fish communities.

In conclusion, omnivorous fish as tilapia, which are often dominant in tropical and subtropical waters, can enhance the growth of phytoplankton and change the size-structure of its assemblages. The effects are obtained by combining manifold strategies: tilapia increases nutrients concentration as well as TSS, may decrease the impact of zooplankton grazers, lowers the competitive effects exerted by periphyton. All these effects synergically deteriorate water quality. Fish removal, although difficult, probably represents an important step in some management plans aimed at improving water quality.

Acknowledgments. This work was supported by the National Natural Science Foundation of China (No. 41771100; 41811530056).

References

Agawin NSR, Duarte CM, Agusti S. 2000. Nutrient and temperature control of the contribution of picoplankton to phytoplankton biomass and production. *Limnol Oceanogr* 45: 591–600.

APHA. 1998. Standard methods for the examination of water and wastewater, 20th ed. Washington: American Public Health Association.

Attayde JL, Hansson LA. 1999. Effects of nutrient recycling by zooplankton and fish on phytoplankton communities. *Oecologia* 121: 47–54.

Attayde JL, Menezes RF. 2008. Effects of fish biomass and planktivore type on plankton communities. *J Plankton Res* 30: 885–892.

Attayde JL, Nes EHV, Araujo AIL, Corso G, Scheffer M. 2010. Omnivory by planktivores stabilizes plankton dynamics, but may either promote or reduce algal biomass. *Ecosystems* 13: 410–420.

Canonico GC, Arthington A, Mccrary JK, Thieme ML. 2005. The effects of introduced tilapias on native biodiversity. *Aquat Conserv: Mar Freshwat Ecosyst* 15: 463–483.

Carpenter S, Kitchell J. 1993. The trophic cascade in lakes. *Biol Conserv* 83: 349–350.

Charvet P, Occhi TVT, Faria L, *et al.* 2021. Tilapia farming threatens Brazil's waters. *Science* 371: 355–356.

Chen B, Liu H. 2010. Relationships between phytoplankton growth and cell size in surface oceans: interactive effects of temperature, nutrients, and grazing. *Limnol Oceanogr* 55: 965–972.

Chen B, Liu Z. 2012. Effect of filter-feeding omnivorous fish on zooplankton community. *Ecolog Sci* 31: 161–166.

Christoffersen K, Riemann B, Klysner A, Søndergaard M. 1993. Potential role of fish predation and natural populations of zooplankton in structuring a plankton community in eutrophic lake water. *Limnol Oceanogr* 38: 561–573.

Cordero PU, Ekvall MK, Hansson LA. 2015. Responses of cyanobacteria to herbivorous zooplankton across predator regimes: who mows the bloom?. *Freshw Biol* 60: 960–972.

Cui DY, Wang JT, Tan LJ, Dong ZY. 2016. Impact of atmospheric wet deposition on phytoplankton community structure in the South China Sea. *Estuar Coast Shelf Sci* 173: 1–8.

Cyr H, Curtis JM. 1999. Zooplankton community size structure and taxonomic composition affects size-selective grazing in natural communities. *Oecologia* 118: 306–315.

Drenner RW, Gallo KL, Baca RM, Smith JD. 1998. Synergistic effects of nutrient loading and omnivorous fish on phytoplankton biomass. *Can J Fish Aquat Sci* 55: 2087–2096.

Figueredo CC, Giani A. 2005. Ecological interactions between Nile tilapia (*Oreochromis niloticus*, L.) and the phytoplanktonic community of the Furnas Reservoir (Brazil). *Freshw Biol* 50: 1391–1403.

Finkel ZV, Beardall J, Flynn KJ, Quigg A, Rees TAV, Raven JA. 2010. Phytoplankton in a changing world: cell size and elemental stoichiometry. *J Plankton Res* 32: 119–137.

Getachew T, Fernado CH. 1989. The food habits of an herbivorous fish (*Oreochromis niloticus* Linn.) in Lake Awasa, Ethiopia. *Hydrobiologia* 174: 195–200.

Guariento RD, Carneiro LS, Caliman A, Bozelli RL, Leal JJF, Esteves FDA. 2010. Interactive effects of omnivorous fish and nutrient loading on net productivity regulation of phytoplankton and periphyton. *Aquat Biol* 10: 273–283.

Gurgel J, Fernando CH. 1994. Fisheries in semi-arid northeast Brazil with special reference to the role of tilapias. *Int Revue ges Hydrobiol* 79: 77–94.

Hansson LA. 1992b. Factors regulating periphytic algal biomass. *Limnol Oceanogr* 37: 322–328.

Hansson LA. 1990. Quantifying the impact of periphytic algae on nutrient availability for phytoplankton. *Freshw Biol* 24: 265–273.

Hansson LA. 1992a. The role of food chain composition and nutrient availability in shaping algal biomass development. *Ecology* 73: 241–247.

He H, Jin H, Jeppesen E, Li K, Liu Z, Zhang Y. 2018. Fish-mediated plankton responses to increased temperature in subtropical aquatic mesocosm ecosystems: implications for lake management. *Water Res* 144: 304–311.

Hilligsøe KM, Richardson K, Bendtsen J, Sørensen LL, Nielsen TG, Lyngsgaard MM. 2011. Linking phytoplankton community size composition with temperature, plankton food web structure and sea-air CO_2 flux. Deep-Sea Res Part I-Oceanogr Res Pap 58: 826–838.

Huang B, Lin X, Hong H. 2000. Distribution and environmental controlling of picophytoplankton in western Xiamen waters. *J Oceanogr Taiwan Strait* 19: 330–336.

Iglesias C, Mazzeo N, Meerhoff M, *et al.* 2011. High predation is of key importance for dominance of small-bodied zooplankton in warm shallow lakes: evidence from lakes, fish exclosures and surface sediments. *Hydrobiologia* 667: 133–147.

Jeppesen E, Jensen JP, Søndergaard M, *et al.* 2004. Impact of fish predation on cladoceran body weight distribution and zooplankton grazing in lakes during winter. *Freshw Biol* 49: 432–447.

Jiménez-Montealegre R, Verdegem M, Zamora JE, Verreth J. 2002. Organic matter sedimentation and resuspension in tilapia (*Oreochromis niloticus*) ponds during a production cycle. *Aquac Eng* 26: 1–12.

Kolding J. 1993. Population dynamics and life-history styles of Nile tilapia, *Oreochromis niloticus*, in Ferguson's Gulf, Lake Turkana, Kenya. *Environ Biol Fishes* 37: 25–46.

Kormas KA, Garametsi V, Nicolaidou A. 2002. Size-fractionated phytoplankton chlorophyll in an Eastern Mediterranean coastal system (Maliakos Gulf, Greece). *Helgoland Mar Res* 56: 125–133.

Kour R, Bhatia S, Sharma KK. 2014. Nile tilapia (*Oreochromis niloticus*) as a successful biological invader in Jammu (J&K) and its impacts on native ecosystem. *Biology* 10: 1–5.

Lazzaro X, Bouvy M, Ribeiro RA, *et al.* 2003. Do fish regulate phytoplankton in shallow eutrophic Northeast Brazilian reservoirs?. *Freshw Biol* 48: 649–668.

Li WKW. 2006. Macroecological patterns of phytoplankton in the northwestern North Atlantic Ocean. *Nature* 419: 154–157.

Lu K, Jin C, Dong S, Gu B, Bowen SH. 2006. Feeding and control of blue-green algal blooms by tilapia (*Oreochromis niloticus*). *Hydrobiologia* 568: 111–120.

Lu K, Jin C, Wang Y. 2005. Control of cyanobacterial blooms in eutrophication lakes by tilapia. *J Fish China* 29: 811–818.

Lynch M, Shapiro J. 1981. Predation, enrichment, and phytoplankton community structure. *Limnol Oceanogr* 26: 86–102.

Marañón E. 2015. Cell size as a key determinant of phytoplankton metabolism and community structure. *Annu Rev Mar Sci* 7: 241–264.

Mayer T. 2020. Interactions of fish, algae, and abiotic factors in a shallow, tropical pond. *Hydrobiologia* 847: 4145–4160.

Meerhoff M, Mello FT, Kruk C, *et al.* 2012. Environmental warming in shallow lakes: A review of potential changes in community structure as evidenced from space-for-time substitution approaches. *Adv Ecol Res* 46: 259–349.

Menezes RF, Attayde JL, Vasconcelos FR. 2010. Effects of omnivorous filter-feeding fish and nutrient enrichment on the plankton community and water transparency of a tropical reservoir. *Freshw Biol* 55: 767–779.

Naselli-Flores L. 2014. Morphological analysis of phytoplankton as a tool to assess ecological state of aquatic ecosystems: the case of Lake Arancio, Sicily, Italy. *Inland Waters* 4: 15–26.

Persson A. 1997. Phosphorus release by fish in relation to external and internal load in a eutrophic lake. *Limnol Oceanogr* 42: 577–583.

Peterson MS, Slack WT, Woodley CM. 2005. The occurrence of non-indigenous Nile tilapia, *Oreochromis niloticus* (linnaeus) in coastal Mississippi, USA: ties to aquaculture and thermal effluent. *Wetlands* 25: 112–121.

Razlutskij V, Mei X, Maisak N, *et al.* 2021. Omnivorous carp (*Carassius gibelio*) increase eutrophication in part by preventing development of large-bodied zooplankton and submerged macrophytes. *Water* 13: 1497.

Rong Y, Tang Y, Ren L, *et al.* 2021. Effects of the filter-feeding benthic bivalve *Corbicula fluminea* on plankton community and water quality in aquatic ecosystems: a mesocosm study. *Water* 13: 1827.

Saha SD, Jana BB. 1998. Control of bloom in a tropical lake: grazing efficiency of some herbivorous fishes. *J Fish Biol* 53: 12–24.

Sanderson SL, Cheer AY, Goodrich JS, Grazlano JD, Callan WT. 2001. Cross flow filtration in suspension-feeding fishes. *Nature* 412: 439–441.

Schindler DW. 1974. Eutrophication and recovery in experimental lakes: Implications for lake management. *Science* 184: 897–899.

Shiomoto A, Fujimoto Y, Mimura N, Sasaki A, Fujita T. 2018. Seasonal variations of chlorophyll a and environmental factors in the coastal area of the Okhotsk Sea, Hokkaido. *Nippon Suisan Gakk* 84: 241–253.

Søndergaard M, Liboriussen L, Pedersen AR, Jeppesen E. 2008. Lake restoration by fish removal: short- and long-term effects in 36 danish lakes. *Ecosystems* 11: 1291–1305.

Starling F, Lazzaro X, Cavalcanti C, Moreira R. 2002. Contribution of omnivorous tilapia to eutrophication of a shallow tropical reservoir: evidence from a fish kill. *Freshw Biol* 47: 2443–2452.

Stockner JG, Antia NJ. 1986. Algal picoplankton from marine and freshwater ecosystems: A multidisciplinary perspective. *Can J Fish Aquat Sci* 43: 2472–2503.

Stockner JG. 1988. Phototrophic Picoplankton: An overview from marine and freshwater ecosystems. *Limnol Oceanogr* 33: 765–775.

Suresh AV, Lin CK. 1992. Tilapia culture in saline waters: A review. *Aquaculture* 106: 201–226.

Tõnno I, Agasild H, Kõiv T, Freiberg R, Nõges P, Nõges T. 2016. Algal diet of small-bodied crustacean zooplankton in a cyanobacteria-dominated eutrophic lake. PLoS ONE 11: e0154526.

Torres GS, Silva LHS, Rangel LM, Attayde JL, Huszar LM. 2015. Cyanobacteria are controlled by omnivorous filter-feeding fish (*Nile tilapia*) in a tropical eutrophic reservoir. *Hydrobiologia* 765: 115–129.

Tudorancea C, Fernando CH, Paggi JC. 1988. Food and feeding ecology of *Oreochromis niloticus* (Linnaeus) juveniles in Lake Awassa, Ethiopia. Arch Hydrobiol Suppl 78: 267–289.

Turker H, Eversole AG, Brune DE. 2003b. Comparative Nile tilapia and silver carp filtration rates of partitioned aquaculture system phytoplankton. *Aquaculture* 220: 449–457.

Turker H, Eversole AG, Brune DE. 2003a. Filtration of green algae and cyanobacteria by *Nile tilapia*, *Oreochromis niloticus*, in the Partitioned Aquaculture System. *Aquaculture* 215: 93–101.

Vadeboncoeur Y, Carpenter LSR. 2001. Whole-lake fertilization effects on distribution of primary production between benthic and pelagic habitats. *Ecology* 82: 1065–1077.

Vakkilainen K, Kairesalo T, Hietala J, *et al.* 2004. Response of zooplankton to nutrient enrichment and fish in shallow lakes: a pan-European mesocosme xperiment. *Freshw Biol* 49: 1619–1632.

Vanni MJ, Layne CD. 1997. Nutrient recycling and herbivory as mechanisms in the“top-down” effect of fish on algae in lakes. *Ecology* 78: 21–40.

Vanni MJ. 2002. Nutrient cycling by animals in freshwater ecosystems. *Annu Rev Ecol Syst* 33: 341–370.

Vicente IST, Fonseca-Alves CE. 2013. Impact of introduced Nile tilapia (*Oreochromis niloticus*) on non-native aquatic ecosystems. *Pak J Biol Sci* 16: 121–126.

Yu J, Xia M, Kong M, He H, Liu Z, Jeppesen E. 2020. A small omnivorous bitterling fish (*Acheilognathus macropterus*) facilitates dominance of cyanobacteria, rotifers and Limnodrilus in an outdoor mesocosm experiment. *Environ Sci Pollut Res* 27: 23862–23870.

Zhang X, Mei X, Gulati RD, Liu Z. 2015. Effects of N and P enrichment on competition between phytoplankton and benthic algae in shallow lakes: a mesocosm study. *Environ Sci Pollut Res* 22: 4418–4424.

Zhang X, Liu Z, Jeppesen E, Taylor WD, Rudstam LG. 2016. Effects of benthic-feeding common carp and filter-feeding silver carp on benthic-pelagic coupling: implications for shallow lake management. *Ecol Eng* 88: 256–264.

Zhang X, Liu Z, Jeppesen E, Taylor WD. 2014. Effects of deposit feeding tubificid worms and filter-feeding bivalves on benthic pelagic coupling: implications for the restoration of eutrophic shallow lakes. *Water Res* 50: 135–146.

Zhang X, Mei X, Gulati RD. 2017. Effects of omnivorous tilapia on water turbidity and primary production dynamics in shallow lakes: implications for ecosystem management. *Rev Fish Biol Fish* 27: 245–254.

Cite this article as: Zhang L, Mei X, Tang Y, Razlutskij V, Peterka J, Taylor WD, Naselli-Flores L, Liu Z, Tong C, Zhang X. 2022. Effects of Nile Tilapia (*Oreochromis niloticus*) on phytoplankton community structure and water quality: a short-term mesocosm study. *Knowl. Manag. Aquat. Ecosyst.*, 423, 11.