# Phytoplankton characterization in a tropical tidal river impacted by a megacity: the case of the Saigon River (Southern Vietnam)

**Truong An Nguyen** ( truong-an.nguyen@univ-grenoble-alpes.fr )
University Grenoble Alpes: Universite Grenoble Alpes

**Thanh-Son Dao**
Ho Chi Minh City University of Technology: Truong Dai hoc Bach khoa Dai hoc Quoc gia Thanh pho Ho Chi Minh

**Emilie Strady**
MIO: Institut Mediterraneen d'Oceanologie

**Tuyet T.N. Nguyen**
Universite Grenoble Alpes

**Joanne Aimé**
Ho Chi Minh City University of Technology: Truong Dai hoc Bach khoa Dai hoc Quoc gia Thanh pho Ho Chi Minh

**Nicolas Gratiot**
IRD: Institut de recherche pour le developpement

**Julien Némery**
Grenoble INP: Institut Polytechnique de Grenoble

---

**Research Article**

**Keywords:** cyanobacteria, microalgae, dominant species, eutrophication, trace metals, urbanization

**Posted Date:** June 2nd, 2021

**DOI:** https://doi.org/10.21203/rs.3.rs-467486/v1



---

**Version of Record:** A version of this preprint was published at Environmental Science and Pollution Research on August 16th, 2021. See the published version at https://doi.org/10.1007/s11356-021-15850-x.

# Abstract

The spatiotemporal variation of phytoplankton and their relationship with environmental variables were analysed in the Saigon River – a tropical river in Southern Vietnam. Two longitudinal profiles were conducted during dry and rainy season at 18 sampling sites covering more than 60 km long in the river. Besides, a bi-weekly monitoring conducted upstream, urban area (Ho Chi Minh City - HCMC) and downstream of Saigon River was organized from December 2016 to November 2017. The major phytoplankton were diatoms (e.g. *Cyclotella* cf. *meneghiniana, Leptocylindrus danicus, Aulacoseira granulata*), cyanobacteria (*Microcystis* spp., *Raphidiopsis raciborskii, Pseudanabaena* sp) and euglenoids (*Trachelomonas volvocina*). Commonly freshwater phytoplankton species and sometimes brackish water species were dominant during the monitoring. Phytoplankton abundances in dry season were much higher than in rainy season (> 100 times) which was explained by a shorter riverine water residence time and higher flushing capacity during the dry season. There was a clear separation of phytoplankton abundance between the urban area and the remaining area of Saigon River because of polluted urban emissions of HCMC. Redundancy analysis shows that the environmental variables (TOC, Nitrogen, pH, salinity, Mo, Mn) were the driving factors related to the dominance of *L. danicus* and *Cyclotella cf. meneghiniana* in the upstream river and urban section of Saigon River. The dominance of cyanobacterium *Microcystis* spp in the downstream of Saigon River was related to higher salinity, Mg, Cu concentrations and lower concentrations of nutrients, Mn, Co, Mo. The dominance of potentially toxic cyanobacteria in Saigon River possess health risk to local residents especially upon the increasing temperature context and nutrient loading into the river in the next decades.


# 1. Introduction

Anthropogenic activities cause water quality degradation that are often accelerated by global warming, and can enhance the eutrophication of water bodies, leading to cyanobacterial mass development and ecological health problems of rivers (Machado et al. 2018, Paerl and Huisman 2009). Phytoplankton, including microalgae and cyanobacteria, are primary producers in aquatic ecosystems. Many environmental parameters can regulate their occurrence and development. In particular, physical factors such as light intensity, temperature, pH, turbidity, water mixing by turbulence and current flow strongly influence phytoplankton development (Marinho and Huszar 2002, Wetzel 2001, Zhang and Prepas 1996). Besides carbon dioxide uptake and oxygen liberation during their photosynthesis, phytoplankton consume nutrients (e.g. nitrate, ammonium and phosphorus), require silicate (in case of diatoms and golden algae), and utilize trace elements (e.g. trace metals) for genetic, physical and biochemical processes in their cells during growth (Cavet et al. 2003, D. Tilman et al. 1986, Granéli and Turner 2006, Wetzel 2001). As all living species (Xu et al. 2020), phytoplankton species have specific environmental niche regarding nutrient concentrations and ratios (e.g. Si :P; N :P), trace element concentrations and physical conditions to ensure their optimal growth (D. Tilman et al. 1986, Sabour et al. 2009, Sivonen 1990, Varol and Şen 2018, Wetzel 2001). Phytoplankton are also food for animals of higher trophic level of the food chain, e.g. zooplankton, planktivorous fishes. Hence, their structure and abundance would

directly and indirectly influence the production of aquatic animals at multitrophic levels and the aquatic community structure (Lehman et al. 2009).

The dynamics, growth, abundance of phytoplankton in temperate freshwater bodies are better documented in temperate regions (e.g. Dokulil and Teubner 2000, Downing et al. 2001, e.g. Elser et al. 1990, Yang et al. 2020) than in tropical regions (e.g. Dao et al. 2016, Soares et al. 2007). In tropical rivers, the studies often focused on only one phytoplankton group such as cyanobacteria (e.g. DC Oliveira et al. 2019, Ferrari et al. 2011, Nguyen et al. 2017) or diatoms (e.g. Bellinger et al. 2006, Dalu and Froneman 2016, Duong et al. 2019, Triest et al. 2012) rather than on both microalgae and cyanobacteria (e.g. Soares et al. 2007, Varol and Şen 2018). The dynamics of phytoplankton in rivers from Southeast Asia are poorly reported and understood (Duong et al. 2019, Hoang et al. 2018) and some environmental parameters like trace metals are rarely considered during the investigations (Wang et al. 2020) .

The Saigon River in Southern Vietnam offers considerable services to more than 8 million inhabitants such as drinking water supplies, agriculture, aquaculture, navigation and recreational purposes (Strady et al. 2017). The provinces in the Saigon River basin (e.g. Ho Chi Minh City, Tay Ninh and Binh Duong Provinces) are fastly developing in terms of economy, industry and urbanization, but those activities contribute to the deterioration of the water quality, mainly by the anthropogenic activities and their release to the river (Lahens et al. 2018, Nguyen et al. 2019a, Strady et al. 2017). In this study, we investigated the spatial and temporal variation of phytoplankton and their relationship with environmental variables in the Saigon River in the context of fast urbanization of Ho Chi Minh City (HCMC). The main objectives are i) to assess the structure of phytoplankton and their diversity in the Saigon River as a tropical estuary ii) to characterize the main environmental drivers explaining the temporal and spatial structure of phytoplankton by statistical approach, and iii) to discuss potential risk of the urban contamination and management issues on the phytoplankton dynamics.

## 2. Material And Methods

# 2.1 Study area

The Saigon River, a part of the Saigon–Dongnai River basin located in Southern Vietnam, is about 250 km long with a catchment area of 4,717 $km^2$ up to the confluence with the Dongnai River (Fig. 1). Upstream of the Saigon River, the Dau Tieng Reservoir (270 $km^2$ and 1,580 million $m^3$) was constructed in 1985 for irrigation and flood protection purposes and to control the intrusion of saline water from East Sea Vietnam (Trieu et al. 2014). Flowing through HCMC, the Saigon River is connected with urban canals and further joins the Dongnai River. The Dongnai River (basin area of 26,449 $km^2$) takes its source from Central Highland of Vietnam and flows southward through the Tri An Reservoir (323 $km^2$, 2,700 million $m^3$) which was built in 1986 for hydroelectric production and water supply for domestic, industrial, and agricultural uses (Dao et al. 2016). Downstream of HCMC, the two rivers confluence and form the Soai Rap Estuary, which flows to the East Sea Vietnam (South China Sea), 20 km north of the Mekong Delta,

and through the Can Gio mangrove coast. The Can Gio mangrove has been recognized as a biosphere reserve by the United Nations Educational, Scientific and Cultural Organization.

The flow direction of the Saigon River is predominantly driven by the asymmetric semi-diurnal tides that lead to the flow current inversion twice a day, roughly in the range from − 1500 to + 1500 $m^3 s^{-1}$ (Camenen et al. 2021). The region falls within a tropical monsoon climate with two distinct seasons. The rainy season is from May to November and the dry season is from December to April (annual mean precipitation of 1800 mm). During the dry season, the net residual discharge is low (a few tens of $m^3 s^{-1}$) and is mainly controlled by the amount of water released from the Dau Tieng Reservoir to flush out salt intrusion. The mean annual discharge for the Saigon River is roughly estimated to be in the range of 50 $m^3 s^{-1}$ and represents about one-twelfth that of the Dongnai River (613 $m^3 s^{-1}$) (Nguyen et al. 2019a). Water from Saigon River is uptaken by the Saigon Water Supply Company at Hoa Phu station, 30 km upstream of HCMC (Fig. 1) for domestic water demand. Land use is very diverse from the north to the south of HCMC: agricultural activities in the northwest and east (e.g., paddy rice and vegetables) and industrial parks dominate in the north, while the east, and the south of the inner city are dominated by urban settlement. Economically, HCMC has grown rapidly over the last 10 years, and was considered as one of the five most dynamic cities in the world (World Economic Forum general assembly, January 2017). In 2017, the population of HCMC was 8.4 million inhabitants and less than 10% of the domestic wastewater were collected and treated before being released directly into the urban canals or the Saigon River (Nguyen et al. 2020a). Due to untreated effluents from residential and industrial areas, the Saigon River water episodically suffers severe phases of eutrophication (e.g., high levels of nutrients and Chlorophyll a concentrations) (Nguyen et al. 2019a). In the next decade, according to HCMC authorities, an ambitious environmental sanitation project aims at building a drainage network inside the city and establishing ten new wastewater treatment plants (Nguyen et al. 2020a, Tran Ngoc et al. 2016).

## 2.2 Sampling and in situ measurements

A bi-weekly monitoring was undertaken at three sites from December 2016 to November 2017 in the Saigon River (Fig. 1) as previously described in Nguyen et al., (2019a). To summarize it, (i) an upstream station on the Saigon River (SG01) representative of the reference water status before HCMC, (ii) a station located in the urban area (SG10) representative of the impact of the megacity and (iii) a downstream station after the confluence of the Saigon and Dongnai Rivers (SG18) that aims at assessing the global impact of HCMC to the estuarine and coastal waters. For each field survey, water sampling was realized at low tide to consider the dominant fluvial flow from upstream to downstream (Nguyen et al. 2019a). Additionally, two longitudinal profiles during the dry and rainy seasons from upstream (SG01) to downstream of HCMC (SG18) were carried out to understand the spatial fluctuation of phytoplankton and its diversity in the Saigon River. The longitudinal profiles were conducted on-boat on the 19th April and the 20th October 2017 starting at SG01 and ending at SG18 as described in Nguyen et al. 2019b (Fig. 1). A GPS was used to mark longitude and latitude values of each point and for calculating the distance (km) point to point, in which 0 km was SG01 and 60 km was SG18 (Fig. 2)

Table 1
Dominant species of phytoplankton based on the spatial and temporal monitoring in Saigon River.

| Stations/ profiles of sampling | Dominant species (percentage composition, dominant index Y) | |
|---|---|---|
| | Dry season | Rainy season |
| Longitudinal profile (total distance 61 km) | *Cyclotella cf. meneghiniana* (72%, 0.72), *Leptocylindrus danicus* (9%, 0.09), *Synedra* sp (6%, 0.06), *Amphiprora* sp (3%, 0.03), *Aulacoseira granulata* (3%, 0.03) | *Trachelomonas volvocina* (21%, 0.2), *Microcystis* spp* (13%, 0.06), *Pseudanabaena* sp (11%, 0.1), *Raphidiopsis raciborskii* (11%, 0.04), *Cyclotella cf. meneghiniana* (6%, 0.05), *Nitzschia cf. palea* (6%, 0.04), *Anabaena* sp (5%, 0.04), *Oscillatoria* sp (4%, 0.04), *Euglena* sp (4%, 0.04), *Scenedesmus acuminatus* (3%, 0.03), *Navicula* sp (2%, 0.02) |
| SG01 (0 km) | *Leptocylindrus danicus* (38%, 0.17), *Aulacoseira granulata* (45%, 0.12) | *Eunotia* sp (10%, 0.06), *Peridinium* sp (60%, 0.15) |
| SG10 (41 km) | *Cyclotella cf. meneghiniana* (70%, 0.70), *Leptocylindrus danicus* (30%, 0.27) | *Cyclotella cf. meneghiniana* (36%, 0.22), *Leptocylindrus danicus* (35%, 0.21), *Trachelomonas volvocina* (24%, 0.09), |
| SG18 (61 km) | *Cyclotella cf. meneghiniana* (68%, 0.49), *Microcystis* spp (13%, 0.06), *Leptocylindrus danicus* (13%, 0.05) | *Microcystis* spp (50%, 0.34), *Pseudanabaena* sp (17%, 0.05), *Raphidiopsis raciborskii* (16%, 0.05) |
| **Microcystis* spp. consisted of three species *Microcystis aeruginosa, Microcystis botrys* and *Microcystis wesenbergii* | | |

For each sample, surface water (0–50 cm) was taken by using a 2.5 L Niskin bottle in the middle of the river either from a bridge or from a boat to immediately measure *in situ* physico-chemical parameters using a multi-parameter probe (WTW 3420®), e.g. temperature, pH, conductivity, salinity. A 2-L polypropylene bottle for phytoplankton identification and counting was sampled and fixed *in situ* with Lugol solution (Sournia, 1978). Water sample was also collected in a 5-L polypropylene recipient and stored in a cooler at 4°C for further trace metal analysis (Cr, Mn, Fe, Co, Ni, Cu, Zn, Mo, Cd, Pb), additionally to Total Suspended Sediment (TSS), nutrients (Total N, Dissolved Inorganic N -DIN-, Total P, Dissolved Inorganic P -DIP- and Dissolved Silica -DSi), Organic Carbon (Dissolved Organic Carbon – DOC and Particulate Organic Carbon – POC), Chlorophyll a (Chl-a), which were partly published in Nguyen et al., (2019a).

# 2.3 Laboratory analysis

# 2.3.1. Phytoplankton identification and counting

Prior to phytoplankton counting, one litter of water sample was settled at least 48 h in the laboratory (Smayda and Sournia 1978). The supernatant (top water) was removed and the settling material including phytoplankton was transferred into a measuring beaker and concentrated to 10–30 mL

depending on the amount of settling material. Phytoplankton were observed at 100–400 x magnification on a microscope (Optika 150) and identified to species or genus levels, based on morphology following the system of Komárek and Anagnostidis(1988, 1999, 2005) for cyanobacteria, Krammer & Lange-Bertalot(1991) for diatoms and other taxonomy books for green algae, golden algae, dinoflagellates and euglenoids. Phytoplankton were enumerated with a Sedgewick Rafter counting cell (volume of 1 mL; PYSER-SGI, England) with total counted number from 400 individuals or more for each sample(Smayda and Sournia 1978) to ensure that samples are statistically representative. An individual of phytoplankton is defined as a single cell, a trichome or a colony of phytoplankton as they commonly occur in nature. It was not possible to convert density into biovolume because of the lack of measuring of dimension of individual cell. Chlorophyll a concentrations were used as a proxy of phytoplankton biomass.

## 2.3.2. Total suspended solid, nutrients, carbon and Chlorophyll a

As reported in Nguyen et al., (2019a), samples were filtered through GF/F filter (Whatman®, 0.7µm pore size, 47mm diameter) to analyze TSS and dissolved nutrients. Chlorophyll a, in terms of phytoplankton biomass, were measured after filtration through a second GF/F filter using acetone (90%) extraction and spectrophotometry measurement (Aminot and Kérouel 2004). Dissolved nutrients ($PO_4^{3-}$ as DIP, $NO_3^-$ + $NH_4^+$ as DIN and, $SiO_2$ as DSi) were analyzed using standard colorimetric methods (American Public Health Association: APHA 1995). Unfiltered waters were used to measure Total N and Total P using persulfate digestion process and standard colorimetric method (APHA 1995). Reproducibility for replicate measurements was better than 5% for all total and dissolved nutrients. A TOC analyzer apparatus (TOC-V Shimadzu, CARE laboratory) was used to determine Dissolved Organic Carbon (DOC) in filtered water samples (Sugimura and Suzuki 1988). Particulate Organic Carbon (POC) was measured on suspended matter retained on a GF/F filter (ignited at 550°C) using combustion in a LECO CS-125 analyzer (EPOC laboratory, France) with precision better than 5%. The Total Organic Carbon (TOC) is the sum of DOC and POC (mg $L^{-1}$).

## 2.3.3. Dissolved metals

In the laboratory, 100 mL of sample was filtered on PTFE filters (0.20 mm Omnipore®) using an acid pre-cleaned filtration unit, 14 mL were immediately transferred into a 15 mL acid cleaned PE tube and was acidified ($HNO_3$ 1%; Trace Metal grade Fisher®) and kept in the fridge at 4°C. Metals (Cr, Co, Ni, Cu, Zn, Mo, Mn, Fe) were measured by Thermo Scientific iCAPQ ICP-MS (Plateforme AETE HydroSciences / OSU OREME, Montpellier, France) using an added on-line internal solution (Sc, Ge, In and Bi) to correct signal drifts. Accuracy and precision were controlled using the certified reference material riverine water SLRS6. Accuracy was of 100%, 103%, 100%, 94%, 95%, 103%, 102%, 97%, for Cr, Mn, Fe, Co, Ni, Cu, Zn, Mo respectively and precision of 1%.

## 2.4. Statistical analysis and data treatment

Multivariate statistical analysis was applied to environmental parameters and phytoplankton abundances to identify spatio-temporal variation and the relationship between these parameters. Non-parametric tests were performed on bi-weekly monitoring dataset to identify spatial differences between sampling sites and seasonal differences between dry and rainy season in Saigon River by Kruskal-Wallis test with a significance level of $p < 0.05$. Hierarchical cluster analysis (HCA) was performed based on Ward's method and Euclidean distance (as a parameter to measure the similarity) to combine the samples with high similarity relationship into a cluster. HCA was applied on the phytoplankton abundance dataset of each longitudinal profile (18 sampling sites during two seasons, see Fig. 1 for the location). All those tests were performed using XLSTAT-Base software (Addinsoft 2019).

The dominant species of phytoplankton (in taxa level) were determined based on equation from Zhao et al., (2017):

$$Y = \left(\frac{n_i}{N}\right) f_i \qquad \text{[Eq. 1]}$$

where $n_i$ is the density of species $i$ in a sampling period, $N$ is the total number of all species, $f_i$ is the frequency of the species $i$ present in the sampling period, $Y > 0.02$ indicates dominant species.

Principal component analysis (PCA) was used to determine the patterns of environmental variables (physical-chemical parameters, nutrients, metals and chlorophyll a) in dry and rainy seasons. The similarities of data samples between sampling sites were also detected by PCA. The redundancy analysis (RDA) was used to identify the main drivers of environmental variables on the phytoplankton community in Saigon River. RDA method has been applied successfully in many studies such as (Cavalcanti et al. 2020, Dhib et al. 2015, Varol and Şen 2018) to define the structuring effects of locations or seasons along with environmental conditions on the phytoplankton abundance. All environmental parameter data were $log(x + 1)$ transformed prior to performing the PCA. In the RDA, phytoplankton abundance data were transformed by hellinger transformation to reduce the influence of high proportion taxa. The multivariate statistical procedures were done by R software version 3.6.3 (2020-02-29) (R Development Core Team) with the help of FactoMineR (Lê et al. 2008), factoextra (Kassambara 2017) and vegan packages (Oksanen et al. 2019).

## 3. Results

# 3.1. Spatial and temporal variation of phytoplankton communities

# 3.1.1. Longitudinal profile

The dominant phytoplankton communities identified along the two longitudinal profiles are presented in Fig. 2. It consisted in mainly species of diatoms (e.g., *Cyclotella* cf. *meneghiniana*, *Leptocylindrus*

*danicus* Cleve 1889, *Synedra* sp, *Amphiprora* sp, *Aulacoseira granulata* (Ehrenberg) Simonsen 1979, *Nitzschia* cf. *palea*; *Navicula* sp), cyanobacteria (e.g., *Raphidiopsis raciborskii* (Woloszynska) Aguilera et al. 2018, *Microcystis* spp, *Pseudanabaena* sp, *Dolichospermum* sp, *Oscillatoria* sp), green algae (e.g, Scenedesmus acuminatus (Lagerheim) Chodat 1902), and euglenoids (e.g., *Euglena* sp, *Trachelomonas volvocina* (Ehrenberg) Ehrenberg 1834) (Table 1).

The longitudinal profiles along the Saigon River revealed contrasted phytoplankton densities, varying from 48,200 to 5,296,000 individuals $L^{-1}$ during the dry season (April 2017; Fig. 2a) and from 1,400 to 35,120 individuals $L^{-1}$ during the rainy season (October 2017; Fig. 2b). During the dry season, the phytoplankton densities showed a similar range of lower densities (< 0.5 million individuals $L^{-1}$) in upstream (SG01-SG04) and downstream (SG11-SG18), but they drastically increased up to around 5.0 million individuals $L^{-1}$ from SG05 to SG10. In contrast, during the rainy season longitudinal profile, there was no sudden increase in phytoplankton densities, but they gradually increased from SG01 to SG14 and decreased in the downstream of Saigon River (Fig. 2b). Beside the significant difference in phytoplankton densities between dry and rainy seasons (lower than 100 times), the dominant species along the Saigon River also differed between the two distinct seasons. The dominant phytoplankton species in dry season longitudinal profile were *Leptocylindrus danicus* (SG01 to SG04, km 0 to km 13) and *Cyclotella* cf. *meneghiniana* (SG05 to SG18, km 17 to km 61) (Fig. 2a, Table 1). The dominant species in rainy season longitudinal profile were *Trachelomonas volvocina* (SG01 to SG12, km 0 to km 45), *Raphidiopsis raciborskii* (SG13 to SG15, km 49 to km 53), and *Microcystis* spp., *Psedudanabaena* sp (SG13 to SG18) (Fig. 2b, Table 1).

The HCA on phytoplankton abundances during the dry season showed that there were three major site groups in which phytoplankton community and densities were closely similar: SG01-SG04, SG05-SG12 and SG13-SG18 (Fig. 3a). It also illustrated a higher similarity of phytoplankton's characteristics between two groups SG01-SG04 and SG05-SG12 than downstream group (SG13-SG18). During the rainy season, three groups were identified: SG01-SG07, SG08-SG12 and SG13-SG18 (Fig, 3b). Unlike HCA for the dry season, the similarity in phytoplankton compositions in rainy season at upstream SG01 stretched to SG07 which located in the urban area of HCMC. The close similarity of phytoplankton's characteristics among the sites SG13-SG18 was comparable between the dry and the rainy seasons and had the most difference with the two groups in upstream and urban area. In general, HCA results show that phytoplankton composition and densities along the Saigon River varied along three sections of Saigon River (i) upstream, (ii) urban area and (iii) downstream of HCMC. The temporal variation of phytoplankton communities at these sections will be analyzed in turn based on bi-weekly monitoring over a year at three representative sites (SG01, SG10 and SG18).

# 3.1.2. Bi-weekly monitoring

The phytoplankton communities at the three sampling sites (SG01; SG10; SG18) are presented in Fig. 4. The dominant phytoplankton species, genera and their densities varied among the sites and were globally characterized by: diatoms, (e.g. *A. granulata, Navicula* spp., *L. danicus*, *Cyclotella* cf. *meneghiniana*),

cyanobacteria (e.g. *Microcystis* sp., *R. raciborskii, Pseudanabaena* sp.) green algae (e.g. *S. acuminatus*), euglenoids, (e.g. *T. volvocina*) and dinoflagellates (e.g. *Peridinium* sp.) (Table 1, Fig. 4). A high correlation between total phytoplankton density and chlorophyll *a* at SG01 ($R^2 = 0.91$) and SG10 ($R^2 = 0.93$) was observed (Fig. 4). The phytoplankton densities varied between sites with several orders of magnitude, from 400–327,500 individuals $L^{-1}$ at SG01, from 9,330–2,733,000 individuals $L^{-1}$ at SG10, and from 1,360–45,700 individuals $L^{-1}$ at SG18 (Fig. 4). At SG01, the densities were elevated in March and April 2017 (mainly *L. danicus* and *A. granulata*) and very low (< 50,000 individuals $L^{-1}$) the rest of the survey (Fig. 4a, Table 1). At SG10, phytoplankton densities were extremely high from January to May 2017 (70% *C. meneghiniana* and 30% *L. danicus*) and lower than 200,000 individuals $L^{-1}$ the remaining months (Fig. 4b, Table 1). At SG18 – downstream of Saigon River, there was no significant difference in phytoplankton density between rainy and dry seasons ($p > 0.1$). Phytoplankton densities were the highest during two periods, from January to March 2017 (mainly *C. meneghiniana*) and from June to November 2017 (mainly *Microcystis* spp), and lower than 10,000 individuals $L^{-1}$ the rest of the year (mainly *L. danicus*, *Pseudanabaena* sp and *R. raciborskii* ) (Fig. 4c, Table 1).

# 3.2. Spatial and temporal variation of environmental parameters

The spatiotemporal variations of physico-chemical parameters, carbon, nutrients and chlorophyll *a* were presented in detail in the study by Nguyen et al (2019a). We here summarize the main patterns of the above parameters and addition of metals for longitudinal profiles and bi-weekly monitoring in the Saigon River.

## 3.2.1. Longitudinal profiles

The environmental parameters fluctuated spatially along the Saigon River from upstream freshwaters to downstream brackish estuarine waters (Supplementary S1). The temperature presented low variation along the two profiles but was higher during the dry season than during the rainy season. The pH increased gradually from SG01 to SG18 and was more acid during the rainy season than during the dry season (p-value < 0.05). Saline water intrusion was observed up to SG04 (about 100 km from estuary mouth) during the dry season while it was null during the rainy season along the whole profile. The magnitude of TSS was higher during the dry season than during the rainy season. Chlorophyll *a* concentration in the dry season increased slightly from SG01 to reach extreme values in SG05-SG09 and returned to very low values in SG13-SG18 in the dry season. The extreme values observed in SG05-SG09 were one order of magnitude higher than those observed during the rainy season where the values were very low in comparison. TOC concentrations increased from SG01 and reached their highest values in SG05-SG12 during both seasons and decreased from SG12 to SG18. Total N and DIN concentrations were higher from SG01 to SG12 than in SG13 to SG18 in both seasons. Total P concentrations were higher in SG01-SG12 before decreasing from SG13 to SG18, especially during the rainy season where the values were significantly greater than during the dry season (p-value < 0.05). The DIP concentrations fluctuated slightly in the same range without clear patterns from SG01 to SG18 during both seasons. DSi

values were rather low in SG01-SG08 but increased locally in SG09-SG12 during the dry season or in SG13-SG18 during the rainy season.

Dissolved trace metals (Cr, Co, Ni, Cu, Zn, Mo) and major elements (Fe and Mn) varied along and between the two longitudinal profiles. During the dry season profile, Cu and Fe were significantly correlated to pH and salinity ($p < 0.05$, Supplementary S1) while during the rainy season, Cr, Co, Ni, Cu, Zn and Fe were significantly correlated to pH. In respect to the groups identified by HCA for each season, during the dry season, Cr, Ni and Zn did not fluctuate spatially, whereas Cu exhibited significantly higher concentrations at SG01-SG04 and Fe, Mn, Mo and Co exhibited significantly lower concentrations at SG05-SG08 (i.e. the zone of maximum phytoplankton abundance) ($p < 0.05$). During the rainy season, Cr and Ni did not also vary along the profile's HCA groups, while the lowest concentrations were measured at SG9-SG12 for Cu and at the downstream group (SG13-SG18), for Co, Zn, Mo, Mn, Fe, the highest Mn and Co being measured at SG09-SG12 (Supplementary S2). The levels of dissolved elements measured during the two profiles are under the level recommendations for drinking water (WHO 2011) and Vietnamese water quality guideline QCVN (QCVN 08:2008/BTNMT).

# 3.2.2. Bi-weekly monitoring

Due to HCMC urban inputs, chlorophyll *a*, TOC, Total P, Total N and, DIN concentrations were significantly (1.5-2 folds) higher in SG10 than in both SG01 and SG18 (p-value < 0.05). The Saigon River's water status was then classified as "bad quality" and eutrophic according to European and Vietnamese standard for water quality (Nguyen et al. 2019a). The pH was acid in SG01 and lower than in the two other sites (p-value < 0.05). DIP was quite low and in the same range in the three sites, while DSi increased slightly from upstream to downstream. Trace metals (Cr, Co, Ni, Cu, Zn, Mo) and major elements (Fe and Mn) evidenced significant spatial differences over the one-year biweekly monitoring ($p < 0.05$) (Table 2). Concentrations of Cr, Co, Ni and Fe were lower at SG18 than at SG01 and SG10, where they were similar. Concentrations of Zn were lower at SG18 and higher at SG01 while they were higher at SG10 for Mn. Concentrations of Mo evidenced higher levels at SG10 than at SG01, SG18, while Cu concentrations were higher at SG01, followed by SG18 and by SG10.

Table 2
Seasonal variation of environmental parameters in the Saigon River and its estuary during rainy and dry periods between December 2016 and November 2017 (mean and SD in bold and min-max values)

| | | **SG01*** **Upstream** | | **SG10*** **Urban area** | | **SG18*** **Downstream** | |
|---|---|---|---|---|---|---|---|
| | | *Dry* | *Rainy* | *Dry* | *Rainy* | *Dry* | *Rainy* |
| **Physico-chemical** | | | | | | | |
| T °C | °C | **29.7 (1.2)** *27.9–31.7* | **30.0 (1.0)** *28.0-31.4* | **29.6 (0.9)** *27.9–31.1* | **30.0 (0.8)** *28.8–31.2* | **29.3 (1.0)** *28.1–31.5* | **29.6 (0.9)** *27.9–30.7* |
| pH | | **6.2 (0.1)** *6.0-6.4* | **6.4 (0.2)** *6.0-6.6* | **6.7 (0.1)** *6.5–6.9* | **6.8 (0.2)** *6.5–7.1* | **6.7 (0.2)** *6.4-7.0* | **6.8 (0.1)** *6.6-7.0* |
| Salinity | ppt | **0 (0)** *0–0* | **0 (0)** 0–0 | **0.5 (0.6)** *0.0-1.6* | **0.0** 0–0 | **1.7 (1.4)** *0.0–4.0* | **0.2 (0.2)** *0-0.5* |
| TSS | mg L-1 | **40.6 (17.9)** *12.1–69.2* | **36.2 (16.5)** *14.9–66.5* | **78.9 (59.0)** *18.7-182.4* | **47.4 (35.6)** *15.2-130.2* | **62.6 (36.8)** *23.3-138.4* | **67.4 (33.6)** *31.2-119.6* |
| **Chl a, carbon and nutrients** | | | | | | | |
| Chl a | $\mu g\ L^{-1}$ | **4.1 (4.8)** *0.3–17.5* | **1.1 (0.7)** *0.3–2.1* | **48.5 (48.3)** *1.8-147.3* | **6.7 (5.8)** *1.0-19.8* | **1.7 (0.8)** *0.5-3* | **1.7 (0.8)** *0.5–3.4* |
| TOC | $mgC\ L^{-1}$ | **6.0 (2.0)** *3.5–10.4* | **4.5 (0.5)** *3.7–5.1* | **8.2 (2.6)** *3.6–13.3* | **6.4 (1.5)** *4.5–9.1* | **3.9 (1.2)** *2.4–7.4* | **3.9 (1.0)** *2.8–6.8* |
| Total P | $mgP\ L^{-1}$ | **0.20 (0.13)** *0.07–0.54* | **0.17 (0.07)** *0.10–0.33* | **0.42 (0.23)** *0.11–0.91* | **0.27 (0.12)** *0.12–0.52* | **0.14 (0.08)** *0.04–0.27* | **0.15 (0.09)** *0.05–0.37* |
| DIP | $mgP\ L^{-1}$ | **0.06 (0.08)** *0.02–0.32* | **0.03 (0.01)** *0.02–0.04* | **0.05 (0.03)** *0.02–0.08* | **0.04 (0.02)** *0.02–0.10* | **0.04 (0.02)** *0.02–0.08* | **0.03 (0.01)** *0.01–0.04* |

*See Fig. 1 for the location of the three sites

| | | SG01* Upstream | | SG10* Urban area | | SG18* Downstream | |
|---|---|---|---|---|---|---|---|
| Total N | mgN $L^{-1}$ | **1.93 (0.58)** *1.18–3.21* | **2.15 (0.39)** *1.54–2.80* | **4.07 (0.99)** *2.84–5.88* | **3.91 (0.61)** *3.07–4.98* | **2.13 (0.76)** *1.24–3.70* | **2.65 (0.68)** *1.07–3.63* |
| DIN | mgN $L^{-1}$ | **0.92 (0.44)** **0.47–1.90** | **0.68 (0.31)** *0.35–1.44* | **1.74 (0.89)** *0.47–3.98* | **1.24 (0.19)** *1.02–1.63* | **1.05 (0.43)** *0.42–1.78* | **0.72 (0.21)** *0.43–0.94* |
| DSi | mgSi $L^{-1}$ | **1.15 (1.02)** *0.05–3.50* | **0.72 (0.14)** *0.47–0.98* | **1.70 (1.57)** *0.12–4.13* | **0.86 (0.56)** *0.23–2.33* | **2.32 (1.71)** *0.37–5.60* | **0.96 (0.39)** *0.56–1.45* |
| **Trace metals** | | | | | | | |
| V | µg $L^{-1}$ | **0.94 (0.14)** *0.73–1.14* | **1.25 (0.15)** *0.94–1.51* | **1.00 (0.29)** *0.41–1.30* | **1.24 (0.41)** *0.49–1.65* | **0.99 (0.14)** *0.78–1.28* | **0.96 (0.12)** *0.76–1.15* |
| Cr | µg $L^{-1}$ | **0.53 (0.47)** *0.19–1.51* | **0.23 (0.16)** *0.16–0.37* | **0.41 (0.11)** *0.19–0.61* | **0.30 (0.26)** *0.19–1.07* | **0.18 (0.16)** *0.06–0.64* | **0.20 (0.13)** *0.10–0.45* |
| Co | µg $L^{-1}$ | **0.49 (0.10)** *0.36–0.68* | **0.45 (0.14)** *0.20–0.73* | **0.59 (0.18)** *0.37–0.97* | **0.47 (0.07)** *0.38–0.61* | **0.19 (0.15)** *0.04–0.56* | **0.10 (0.05)** *0.06–0.12* |
| Ni | µg $L^{-1}$ | **1.70 (0.35)** *1.21–2.49* | **1.94 (1.55)** *1.26–6.54* | **2.00 (0.38)** *1.43–2.68* | **1.75 (0.55)** *1.45–3.37* | **1.55 (0.63)** *0.90–2.59* | **1.21 (0.22)** *1.00-1.68* |
| Cu | µg $L^{-1}$ | **1.40 (0.24)** *1.12–2.06* | **1.56 (0.29)** *1.02–2.16* | **0.70 (0.19)** *0.46–1.06* | **0.76 (0.31)** *0.30–1.12* | **0.85 (0.22)** *0.66–1.47* | **1.07 (0.23)** *0.63–1.29* |

*See Fig. 1 for the location of the three sites

| | | SG01* Upstream | | SG10* Urban area | | SG18* Downstream | |
|---|---|---|---|---|---|---|---|
| Zn | µg $L^{-1}$ | **20.8 (7.9)** *13.8–43.2* | **14.1 (4.1)** *1.0-21.4* | **9.2 (4.00)** *5.7–16.5* | **7.4 (3.3)** *3.8–15.1* | **5.2 (2.2)** *1.5–8.8* | **6.8 (2.5)** *3.2–10.9* |
| As | µg $L^{-1}$ | **0.52 (0.04)** *0.47–0.58* | **0.55 (0.04)** *0.48–0.63* | **0.98 (0.14)** *0.80–1.22* | **1.01 (0.14)** *0.85–1.28* | **0.78 (0.16)** *0.50–0.98* | **0.61 (0.07)** *0.52–0.73* |
| Mo | µg $L^{-1}$ | **0.41 (0.29)** *0.18–1.30* | **0.24 (0.05)** *0.18–0.32* | **0.77 (0.35)** *0.42–1.38* | **0.55 (0.08)** *0.36–0.71* | **0.69 (0.33)** *0.24–1.12* | **0.33 (0.14)** 0.20–0.50 |
| Cd | µg $L^{-1}$ | **0.04 (0.08)** *0.01–0.31* | **0.01 (0.00)** *0.00-0.01* | **0.06 (0.11)** *0.01–0.40* | **0.01 (0.01)** *0-0.03* | **0.01 (0.01)** *0-0.03* | 0.01 **(0.01)** *0-0.03* |
| Pb | µg $L^{-1}$ | **0.09 (0.04)** *0.05–0.18* | **0.06 (0.04)** *0.04–0.07* | **0.14 (0.21)** *0.04–0.81* | **0.06 (0.03)** *0.03–0.15* | **0.03 (0.01)** *0.02–0.05* | **0.10 (0.09)** *0.03–0.31* |
| Mn | µg $L^{-1}$ | **44,4 (9.1)** *28.1–58.5* | **39.5 (11.6)** *20.8–61.9* | **166 (70)** *86–314* | **109 (29)** *68–172* | **55.0 (45.7)** *4.5-121.5* | **23.1 (8.3)** *18.9–29.9* |
| Fe | µg $L^{-1}$ | **24.5 (7.9)** *14.5–44.4* | **23.8 (12.6)** *10.4–52.5* | **21.1 (15.9)** *6.3–63.1* | **20.0 (7.5)** *10.5–29.7* | **4.5 (2.7)** *1.2–8.1* | **23.9 (36.8)** *5.6-130.8* |
| Mg | µg $L^{-1}$ | **1224 (274)** *851–1942* | **1078 (118)** *805–1237* | **4555 (4415)** *1515–14569* | **2097 (522)** *1483–3367* | **14092 (12951)** *2401–35656* | **9211 (10047)** *1737–21382* |

*See Fig. 1 for the location of the three sites

Significant temporal variations between the dry and the rainy seasons were observed for salinity at SG18 and at SG01, SG10 with higher values during the dry season (p-value < 0.05) (Table 2, Supplementary S2). Temperature, pH and, TSS presented similar values during both seasons. Chlorophyll a was one order of

magnitude higher in SG10 during the dry season than in the rainy season ($p < 0.05$) while the difference was much less in SG01 or even null in SG18. As already established in Nguyen et al., (2019a), the carbon and nutrient concentrations did not show a strong variation between the dry and the rainy season. The only noticeable variations were observed in SG01 and SG10, which had significantly greater TOC concentrations during the dry season than during the rainy season (p-value < 0.05). Significant temporal variations between the dry and the rainy were also observed for Mo at SG18 and Mn at SG10 with higher values during the dry seasons and for Fe and Cu at SG18 during the rainy seasons ($p < 0.05$). The other elements (Cr, Co, Ni, Zn) and sites presented similar concentrations during both seasons. The levels of dissolved elements measured during the monitoring were under the level recommendations for drinking water (WHO 2011) and Vietnamese water quality guideline QCVN (QCVN 08:2008/BTNMT).

## 4. Discussion

# 4.1. Effects of spatial and seasonal variation on phytoplankton communities

In large worldwide rivers, the dominant algal groups are diatoms and green algae (Reynolds 2006, Wehr and Descy 1998). In tropical rivers, the main phytoplankton groups observed are diatoms, green algae and cyanobacteria, as for example in the Pomba River, Southeast Brazil (Soares et al. 2007), and in the Tigris River, Turkey (Varol and Şen 2018). Base on Redfield stoichiometric ratio, DSi was not a limiting factor for algal growth in the Saigon River (Nguyen et al. 2019a). The adequate DSi (mean values > 0.6 mg $L^{-1}$; Table 2) for diatoms development (Reynolds 2006) would be an advantage for the high abundance of diatoms, especially in the dry season with a long residence time (Ferreira et al. 2005). However, during the rainy season in the Saigon River, there was the appearance and dominance of cyanobacteria, especially in downstream of the river which suggested the spatial and seasonal effects on phytoplankton communities in Saigon River.

# 4.1.1. Spatial effect on phytoplankton communities

In the longitudinal profiles along the Saigon River (SG01 - SG18) and during the bi-weekly monitoring (SG01. SG10 and SG18), we found significant difference of phytoplankton's characteristics between the urban area and the upstream and downstream parts of HCMC (Fig. 3), suggesting influence of localized environmental factors. The phytoplankton abundance in the urban area (Fig. 2a) were much higher than in upstream and downstream of Saigon River. The phytoplankton abundances at SG01 and SG18 (Fig. 4a, 4c) were within the range of the phytoplankton abundances in the Vam Co River (an affluent of the downstream Saigon River) 920–383,600 individuals $L^{-1}$ (Dao and Bui 2016). During the dry season longitudinal profile, the diatom *L. danicus*, a brackish water species, was dominated at the first four sites, SG01-SG04, which could be related to the slight increase of salinity up to SG01 observed at these stations (Supplementary S1) and could support the conditions for the occurrence and out competition of this diatom to other phytoplankton. The freshwater species (except *L. danicus* – a brackish diatom)

dominated during the dry period in March and April 2017 at the site SG01, which is coherent with the freshwater characteristic (salinity ~ 0) nearly all year around at the site SG01 (Table 2). The abundance of phytoplankton at SG10 during the dry season was around 10 and 100 times higher than that at SG01 and SG18, respectively (Fig. 4). These results were comparable with the phytoplankton abundance in the Tigris and Paraguay Rivers which have been also under effect of anthropogenic pressures (Domitrovic 2002, Varol and Şen 2018). At SG10, the dominance of *Cyclotella* cf. *meneghiniana* and *L. danicus* could be driven by environmental parameters such as salinity, trace elements (e.g. Mo, Mn) and high concentrations of nutrients from urban discharge. Dao and Bui (2016) also observed the dominance of diatoms (*Cyclotella* spp. and *Eunotia* spp) in the surface waters of Vam Co River (15 km from Saigon River Estuary mouth), which is in accordance with the dominance of diatoms species (e.g. *Navicula* sp., *A. granulata, Cyclotella* cf. *meneghiniana*, and *L. danicus)* in our study (Table 1).

Unlike upstream and urban area of HCMC, phytoplankton abundance was quite low in the downstream of HCMC (Fig. 2) and dominated by both diatoms (*Cyclotella* cf. *meneghiniana*) and cyanobacteria (e.g. *Microcystis* spp, *Pseudanabaena* sp, *Raphidiopsis raciborskii*) (Table 1). The site SG16 is the meeting point of the Saigon and Dongnai Rivers (Fig. 1), and the water discharge of the Dongnai River is around 12 times higher than the one of the Saigon River (Nguyen et al. 2019a). The phytoplankton community and abundance would thus be much more influenced by the water from the Dongnai River than by water from the Saigon River at site SG18. Proliferation and bloom forming of cyanobacteria have been commonly observed in the Tri An Reservoir, upstream of the Dongnai River (Dao et al. 2016, Nguyen et al. 2020b). We found that there was a high similarity of cyanobacteria groups (*Microcystis* spp) found in downstream HCMC and at Tri An Reservoir (Dao et al. 2016). Therefore, the commonly presence and dominance of cyanobacteria (Table 1) especially at SG18 (Fig. 4c) could result of (i) the consequence of the diffusion of waters enriched with cyanobacteria, from Tri An Reservoir (Nguyen et al. 2020b); and/or (ii) the saline intrusion and anthropogenic emission which enhanced the cyanobacterial development in the most active estuarine section (Paerl and Huisman 2009).

# 4.1.2. Seasonal effect on phytoplankton abundance

Beside the influence of localized environmental factors, seasonal effects lead to a significant change in phytoplankton communities in tropical estuaries (Bledsoe et al. 2004, Cavalcanti et al. 2020, van Chu et al. 2014). The results of bi-weekly monitoring show that there was a clear difference in the abundance and phytoplankton species between the dry and rainy seasons in the Saigon River (Table 1, Fig. 4). In contrast to the rainy season, the residual water discharge in Saigon River was very low (30 $m^3s^{-1}$) during the dry season (Camenen et al. 2021). The residence time of the water body in this urban section of Saigon River was estimated to be around two months (Nguyen et al. 2021). High load of nutrients from HCMC and long residence time may strongly enhance the phytoplankton development during the dry season in the urban section of the Saigon River. Ferreira et al. (2005) found that higher phytoplankton biomass is more prevalent in estuaries with long residence time even without anthropogenic impact. However, during the rainy season, stronger water discharge from upstream of the Saigon River had shorten residence time of water body. This leads to a decrease of phytoplankton development before

being flushed out of the estuary during the rainy season. The seasonal variation in phytoplankton abundance in the Saigon River has also been found in other tropical estuaries such as Bach Dang Estuary (Vietnam) (van Chu et al. 2014), Red River (Vietnam) (Duong et al. 2019) and Paciencia River Estuary (Brazil) (Cavalcanti et al. 2020). During the dry season, *L. danicus* and *A. granulata* were dominant in upstream of HCMC (SG01), while the freshwater algae *Peridinium* was dominant there in rainy season. At SG10, there was the dominance of another freshwater species (*T. volvocina*) beside the dominance of *Cyclotella* cf. *meneghiniana* and *L. danicus* (Table 1; Figs. 2b, 4b). The appearance of freshwater algae could be due to prevention of saline intrusion supported by the high discharge in the rainy season. This could be also related to the Saigon River morphometry, very deep at this location (> 12 m), and lower water current locally, freshwater algae from inland, and eutrophic canals and creeks connecting to the river.

Besides, during the rainy season longitudinal profile, the similarity of phytoplankton characteristics among the sites did not have a clear separation as in dry season (Fig. 3). This may be linked with the higher water discharge from upstream of Saigon River during the rainy season resulting in the influence of inland water down to urban area. The supporting evidences for this hypothesis are that (i) there was not a significant increase of phytoplankton abundance along Saigon River (max of 35,120 ind $L^{-1}$), and (ii) typical inland algae (*T. volvocina*; Reynolds, 2006) dominated during the rainy season longitudinal profile. Besides, the dominance of the freshwater species of *T. volvocina* and *Pseudanabaena* sp. at the site urban section of Saigon River by the end of the rainy season (Oct, Nov 2017) is supported by the dominance of *T. volvocina* at 12 sites during the rainy season (Fig. 2b; Table 1). The euglenoids *Trachelomonas* and dinoflagellates *Peridinium* are typical genera of nutrient enrich trophic lakes (Reynolds 2006) such as the ones observed in the Saigon River.

Another noteworthy difference is the formation of cyanobacterial species during the rainy season. Bi-weekly monitoring and longitudinal profile during rainy season both detected cyanobacterium *Microcystis* spp and *R. raciborskii* in the downstream of the Saigon River. While *Microcystis* spp could have originated from Tri An Reservoir (Dao et al. 2016), Cyanobacterium *R. raciborskii* may be related to the high TSS concentrations in the downstream area during the rainy season (Table 2). Some cyanobacterial species such as *R. raciborskii* and *Microcystis* spp. have air aerotopes in their cells which give them a buoyant capacity to surface of water (Mhlanga et al. 2006). The cyanobacterium *R. raciborskii* has a filamentous form which helps to capture more light in turbid waters (Reynolds 2006). Therefore, the species *R. raciborskii* could withstand the low radiation in water column in high TSS condition and maintain their normal photosynthesis activities in top water.

# 4.2. Relationship between phytoplankton and environmental factors

# 4.2.1. Influence of urban inputs on the environmental variables and phytoplankton biomass

The PCA was used to identify the variation patterns of environmental parameters and similarities between three sections of the Saigon River. Based on the high correlation of Chl-*a* and phytoplankton abundances ($R^2$ = 0.9, Fig. 4), the PCA results could be used to identify the impact of urban discharge on the phytoplankton biomass (i.e. described by Chl-*a* concentration). As shown in Table 2, there were differences in concentrations of environmental parameters (including DIN, TN, TP, TOC, Mo, Mn, Cu and Ch-a) between urban area and upstream, downstream parts of Saigon River. Similar to the HCA results on phytoplankton abundances along the Saigon River, the PCA results on environmental parameters shows a clear separation of the three monitoring locations (SG01, SG10 and SG18) (Fig. 5). The PCA results showed that the water quality in SG10 suffered from the most diverse environmental parameters (TSS, TOC, nutrients, Chl-*a* and dissolved metals), which were mainly related to pollution from urban area of HCMC. During the dry season, the first principal component (PC1) mainly represented metal, pH and salinity variables mainly linked to SG18 (downstream section) (Fig. 5a). In PC2 of the dry season (Fig. 5a) and PC1 of the rainy season (Fig. 5b), the most important variables were the nutrients, TOC, Chl-a, which associated with SG10 (urban section of the Saigon River). This illustrated that the urban area was affected by urban discharges containing high pollutant concentrations from the connecting canals and creeks during both seasons.

# 4.2.2. Main driving factors of phytoplankton development

RDA method allowed to evaluate the influence of environmental variables on the phytoplankton composition and abundances (Cavalcanti et al. 2020, Varol and Şen 2018). Based on RDA, 18 environmental parameters can explain 72% ($R^2$ adjusted = 0.35) and 68% ($R^2$ adjusted = 0.31) of variation of phytoplankton abundances during dry and rainy season, respectively. Only abundances of *L. danicus*, *Cyclotella* cf. *meneghiniana*, *Microcystis* spp, *Eunotia* sp had a clear relationship with environmental parameters while other dominant species (mainly in rainy season) such as *Peridinium* sp, *A. granulata*, *T. volvocina*, *R. raciborskii, Pseudanabaena* sp have not been explained by RDA.

During the dry season (Fig. 6a), the abundance of *Cyclotella* cf. *meneghiniana* was related with pH, salinity, TN, DIN, Chl-a, Mn, Mg and Mo. This result coincides with the positive relationship of Chl-a, TN, Mg and Mo in SG10 (Fig. 5a; Table 2). This indicates that the majority of *Cyclotella* cf. *meneghiniana* abundance contributed to the biomass of Chl-a measured at the urban area of the Saigon River. While Mn plays an important role in the photosynthesis, Mg is an essential element in algal cellular proliferation which reduces the loss of chlorophyll pigments (Bogorad 1966). Therefore, the increase of Mn and Mg in the urban area could support the growth of phytoplankton in the Saigon River. However, according to the longitudinal profile during the dry season, there was no difference in Mn, Mg concentrations between at SG04 and SG05 (Supplement 1), while diatom abundance suddenly increased from about 500,000 to 4,000,000 ind $L^{-1}$. *Cyclotella* cf. *meneghiniana* became the highest densely phytoplankton, replacing *L. danicus* at SG05. Therefore, the metals (Mn and Mg) were not factors leading to the predominance of *Cyclotella* cf. *meneghiniana* in the Saigon river. Relationship between *Cyclotella* cf. *meneghiniana* and salinity clarified this phenomenon. The salinity intrusion of the Saigon River during the dry season was about 90–100 km from the estuary mouth (SG05-SG06), the salinity reached a maximum of about 0.5

during the dry season at SG05. Therefore, the salinity at SG04 was rarely greater than zero. This interesting finding showed that freshwater diatom *Cyclotella* cf. *meneghiniana* can also dominate in non-zero salinity areas (urban area and downstream section). The second dominant diatom in the Saigon River, *L. danicus* was not associated with nutrients but with TOC and inversely correlated with DSi and Mg. The silicate is a macronutrient for diatoms because it is the main component of the frustules of diatoms (Wetzel 2001). However, it is out of our expectation that there was no correlation between diatoms and DSi. The sufficient levels of silicate in the Saigon River (> 0.5 mg $L^{-1}$; Table 2) helps to explain the lack of this relationship (Nguyen et al. 2019a). Besides that, the increase of Mg concentration from SG01 to SG18 (1224 to 14092 µg $L^{-1}$) (Table 2) has led *Cyclotella* cf. *meneghiniana* outgrown *L. danicus* which fell sharply at SG18. *Leptocylindrus danicus* and TOC were positively linked in this study because high phytoplankton abundance contributed to the TOC in surface water, as previously stated in Nguyen et al (2020a). In water, pH regulates the bioavailability of trace metals for phytoplankton, and influences on the rate of phosphate uptake in cyanobacteria (Healey 1982). pH in water would increases upon mass development of phytoplankton (Shapiro 1990). Some phytoplankton species could not grow at very low pH (below 4; (Whitton 1992)) or high pH (greater than 8.6; (Wetzel 2001)). The range of pH in our study (6.2–6.8; Table 2) would be favourable for the phytoplankton development (e.g. *Cyclotella* cf. *meneghiniana*).

During the rainy season, RDA results generally show that diatom densities (*Cyclotella* cf. *meneghiniana*, *L. danicus*) were positively related to variance of TOC, nutrients (DIN, TN, DIP) and metals (Mn, Mo, Co). The positive relationship of phytoplankton divisions with nitrogen and phosphorus compounds in this study is supported by previous investigations (Varol and Şen 2018, Yang et al. 2020). The abundance of *Eunotia* sp. during the rainy season was highly related with the Cu, Zn, Co concentrations but negatively related with pH. This species might be sensitive lower concentration of pH in SG01 where the pH was about 6.2–6.4 (Table 2). This was in contrast to *Cyclotella* cf. *meneghiniana* or *L. danicus* that dominated at SG10 (pH range 6.7–6.8). This result shows that the small difference in pH between upstream (SG01) and urban area (SG10) may result in the change of dominant species along the Saigon River.

The participation of trace elements such as metals were also able to prevent or stimulate the phytoplankton development (Duong et al. 2006, Morin et al. 2007, Rochelle-Newall et al. 2011, Twining and Baines 2013). Trace metals have very important roles in photosynthesis and assimilation of macronutrients (Granéli and Turner 2006). For example, Co occurs in the enzymes, nitrate reductase and nitrogenase, hence needed for nitrate assimilation and nitrogen fixation; Mn occurs in superoxide dismutase, an antioxidant enzyme in cells; Zn has a contribution in DNA transcript and is needed to acquire phosphorus from organic phosphate esters; and Co has a unique requirement in vitamin $B_{12}$ (Granéli and Turner 2006). Cu at low concentrations is essential element for algal growth (Granéli and Turner 2006), but would be toxic to algae at high concentration (e.g. some dozen µg $L^{-1}$). These statements are consistent with the results in this study except for the case of *Microcystis* spp found during the rainy season. The abundances of this cyanobacteria were mostly positive related with salinity,

Mg and inversely related with metals such as Co, Zn, Cu. Our results showed Cu was at very low concentration (Table 2) hence it remains unclear why phytoplankton divisions of cyanobacteria, green algae and diatoms negatively linked with Cu. The other dominant species during the rainy season have no clear link with environmental parameters in the Saigon River. These species may be affected by the hydrological regime (e.g. stronger flushing capacity) and the intrusion of species from Tri An Reservoir rather than the formation and development of phytoplankton within Saigon River.

## 4.3. Potential risks and management issue

Surface water of the Saigon River especially at the area closed to the urban area is under eutrophic status (Nguyen et al., 2019a). Cyanobacterial species were observed in the river during the monitoring period (e.g. SG13-SG18 during the dry and rainy season longitudinal profile, Fig. 2b; during April–August at SG18 of the bi-weekly monitoring, Figs. 4c). Hence, anthropogenic emission and climate context in the future could induce a possible increase of cyanobacterial blooms and then be a threat for the water resources (Paerl and Huisman 2009). Toxic cyanobacterial species such as *Microcystis aeruginosa*, and *Raphidiopsis. raciborskii* and their toxin production (microcystin and cylindrospermopsin) in surface water exceeding the WHO safety guidelines (1 μg $L^{-1}$) were reported in the Dau Tieng Reservoir – upstream the Saigon River (Pham et al. 2017). Surface water from the Saigon River is used for domestic use but the water treatment system for drinking water supplies in Southern Vietnam could not totally remove the dissolved cyanobacterial toxins. Therefore, the presence and dominance of potential toxic cyanobacteria (e.g. *M. aeruginosa, R. raciborskii*) points out some potential health risks to local residents. This risk is likely to be exacerbated in the near future with an increase of domestic nutrient inputs due to the expected growth of the total population in HCMC by 200% in 2040 (Nguyen et al. 2020a). The local authorities have already planned to upgrade the wastewater treatment capacities to fit with this scenario. The realization of such program is challenging and will deal with the management of both rapid urban development and water resource restauration and protection.

# 5. Conclusions

During the spatiotemporal monitoring in the Saigon River, we found common phytoplankton divisions characterized for running waters such as diatoms, cyanobacteria, green algae, dinoflagellates and euglenoids. During the monitoring period, most of dominant species were freshwater species (e.g., *A. granulata, Navicula* sp., *S. acuminatus, Microcystis* spp., *R. raciborskii*) but some of them were brackish water species (e.g. *L. danicus*,) which are all worldwide distribution phytoplankton. From the upstream to the urban area of Saigon River (SG01-SG12), the two major phytoplankton divisions in the longitudinal profiles were representatives of diatoms (*Cyclotella* cf. *meneghiniana, L. danicus*) during the dry season, and appearance of dinoflagellates (*Peridinium* sp), euglenoids (*T. volvocina*) during the rainy season. Cyanobacterial species appeared and were dominant downstream of the river, from SG13-SG18 during the rainy season. During the temporal monitoring at the three sites (SG01-upstream, SG10-urban, SG18-downstream of Saigon River), the dominant species at SG01 were diatoms (*A. granulata, L. danicus)*, euglenoids (*Eunotia* sp) and dinoflagellates (*Peridinium* sp) while dominant species at SG10 were mainly

diatoms (*Cyclotella* cf. *meneghiniana*), and dominant species at SG18 were both diatoms (*Cyclotella* cf. *meneghiniana*) and cyanobacteria (*Microcystis* spp, *R. raciborskii*).

Phytoplankton pattern in the Saigon River varied seasonally, with clear separation between the urban area and the remaining area of the monitored river during the dry season and more similarity from the upstream to the urban area during the rainy season. The abundance of phytoplankton was much higher during the dry season period in the urban area, from 10 to 100 folds higher than the upstream and downstream of the Saigon River, respectively. The mass phytoplankton proliferation in urban area could be linked to the low residual discharge in the Saigon River and nutrient inputs from canal and creeks, as megacity emission of HCMC. Based on RDA results, phytoplankton abundance was linked with pH, salinity, TOC, nitrogen concentration and some metals such as Co, Mo, Cu, Mn. The diatoms abundance did not link with dissolved silicate concentrations in the river because of the redundancy of DSi in Saigon River. The occurrence and dominance of potential toxic cyanobacteria (*Microcystis* spp., *R. raciborskii*) in the downstream part of the river could induce health risk to local residents especially upon the increasing temperature context and nutrient loading into the river modelled for the next decades.

# Declarations

**Acknowledgements:** We acknowledge the CARE and FERN laboratories for providing analytical and field trip facilities.

**Funding information:** This study was conducted at the International Joint Laboratory Le CARE and co-funded by the CMIRA “Saigon River: la ville et fleuve” Region Auvergne Rhone Alpes project (2016-2017) and by the NUTRIM project EC2CO Bioheffect Structurante Initiative (2017-2018).

**Authors’ contributions:** Conceptualization: An Truong Nguyen, Thanh-Son Dao, Julien Némery; Methodology: An Truong Nguyen, Thanh-Son Dao, Emilie Strady, Tuyet T.N. Nguyen, Joanne Aimé, Julien Némery; Formal analysis and investigation: An Truong Nguyen, Thanh-Son Dao, Tuyet T.N. Nguyen, Julien Némery; Writing - original draft preparation: Thanh-Son Dao; Writing - review and editing: An Truong Nguyen, Thanh-Son Dao, Emilie Strady, Nicolas Gratiot, Julien Némery; Funding acquisition: Julien Némery; Supervision: Julien Némery.

**Data availability:** The datasets used and/or analyzed during the current study are available from the corresponding author on reasonable request.

**Competing interests:** The authors declare that they have no competing interests.

**Ethics approval and consent to participate:** Not applicable.

**Consent to publish:** Not applicable.

# References

1. Addinsoft A (2019) XLSTAT statistical and data analysis solution. Long Island, NY, USA
2. Aminot A, Kérouel R (2004) Hydrologie des écosystèmes marins: paramètres et analyses. Editions Quae
3. APHA (1995) Standard methods for the examination of water and wastewater. American Public health Association 17
4. Bellinger BJ, Cocquyt C, O'Reilly CM (2006) Benthic diatoms as indicators of eutrophication in tropical streams. Hydrobiologia 573(1):75–87. doi:10.1007/s10750-006-0262-5
5. Bledsoe EL, Phlips EJ, Jett CE, Donnelly KA (2004) The relationships among phytoplankton biomass, nutrient loading and hydrodynamics in an inner-shelf estuary. Ophelia 58(1):29–47. doi:10.1080/00785236.2004.10410211
6. Bogorad L (1966) The biosynthesis of chlorophylls. In: The chlorophylls. Elsevier, pp 481–510
7. Camenen B, Gratiot N, Cohard J-AA, Gard F, Tran VQ, Nguyen A-TT, Dramais G, van Emmerik T, Némery J (2021) Monitoring discharge in a tidal river using water level observations: Application to the Saigon River. Vietnam Sci Total Environ 761(xxxx):143195. doi:10.1016/j.scitotenv.2020.143195
8. Cavalcanti LF, Cutrim MVJ, Lourenço CB, Sá, Ana Karoline DS, S, Oliveira ALL, de Azevedo-Cutrim, Andrea CG G (2020) Patterns of phytoplankton structure in response to environmental gradients in a macrotidal estuary of the Equatorial Margin (Atlantic coast, Brazil). Estuar. Coast. Shelf Sci. 245(March 2019):106969. doi: 10.1016/j.ecss.2020.106969
9. Cavet JS, Borrelly GPM, Robinson NJ (2003) Zn, Cu and Co in cyanobacteria: selective control of metal availability. FEMS Microbiol Rev 27(2–3):165–181. doi:10.1016/s0168-6445(03)00050-0
10. Tilman D, Kiesling R, Sterner R, Kilham SS, Johnson FA (1986) Green, bluegreen and diatom algae: taxonomic differences in competitive ability for phosphorus, silicon and nitrogen. Archiv Fur Hydrobiologie 106:473–485
11. Dalu T, Froneman PW (2016) Diatom-based water quality monitoring in southern Africa: challenges and future prospects. Water SA 42(4):551. doi:10.4314/wsa.v42i4.05
12. Dao T-S, Bui T-N-P (2016) Phytoplankton from Vam Co River in Southern Vietnam. Environmental Management Sustainable Development 5(1):113. doi:10.5296/emsd.v5i1.8775
13. Dao T-S, Nimptsch J, Wiegand C (2016) Dynamics of cyanobacteria and cyanobacterial toxins and their correlation with environmental parameters in Tri An Reservoir, Vietnam. J Water Health 14(4):699–712. doi:10.2166/wh.2016.257
14. DC Oliveira E, Castelo-Branco R, Silva L, Silva N, Azevedo J, Vasconcelos V, Faustino S, Cunha A (2019) First Detection of Microcystin-LR in the Amazon River at the Drinking Water Treatment Plant of the Municipality of Macapá, Brazil. Toxins 11(11):669. doi:10.3390/toxins11110669
15. Dhib A, Fertouna-Bellakhal M, Turki S, Aleya L (2015) Harmful planktonic and epiphytic microalgae in a Mediterranean Lagoon: The contribution of the macrophyte Ruppia cirrhosa to microalgae dissemination. Harmful Algae 45:1–13. doi:10.1016/j.hal.2015.03.002

16. Dokulil MT, Teubner K (2000) Cyanobacterial dominance in lakes. Hydrobiologia 438(1/3):1–12. doi:10.1023/a:1004155810302
17. Domitrovic YZ de (2002) Structure and variation of the Paraguay River phytoplankton in two periods of its hydrological cycle. Hydrobiologia 472(1):177–196
18. Downing JA, Watson SB, McCauley E (2001) Predicting Cyanobacteria dominance in lakes. Can J Fish Aquat Sci 58(10):1905–1908. doi:10.1139/f01-143
19. Duong TT, Coste M, Feurtet-Mazel A, Dang DK, Gold C, Park YS, Boudou A (2006) Impact of Urban Pollution from the Hanoi Area on Benthic Diatom Communities Collected from the Red, Nhue and Tolich Rivers (Vietnam). Hydrobiologia 563(1):201–216. doi:10.1007/s10750-005-0005-z
20. Duong TT, Hoang TTH, Nguyen TK, Le TPQ, Le ND, Dang DK, Lu XXI, Bui MH, Trinh QH, van Dinh TH, Pham TD, Rochelle-Newall E (2019) Factors structuring phytoplankton community in a large tropical river: Case study in the Red River (Vietnam). Limnologica 76(March):82–93. doi:10.1016/j.limno.2019.04.003
21. Elser JJ, Marzolf ER, Goldman CR (1990) Phosphorus and Nitrogen Limitation of Phytoplankton Growth in the Freshwaters of North America: A Review and Critique of Experimental Enrichments. Can J Fish Aquat Sci 47(7):1468–1477. doi:10.1139/f90-165
22. Ferrari G, del Carmen Perez M, Dabezies M, Miguez D, Saizar C (2011) Planktic Cyanobacteria in the Lower Uruguay River, South America. Fottea 11(1):225–234. doi:10.5507/fot.2011.021
23. Ferreira JG, Wolff WJ, Simas TC, Bricker SB (2005) Does biodiversity of estuarine phytoplankton depend on hydrology? Ecol Model 187(4):513–523. doi:10.1016/j.ecolmodel.2005.03.013
24. Granéli E, Turner JT (2006) Ecology of harmful algae. Ecological Studies. http://dx.doi.org/10.1007/978-3-540-32210-8_1
25. Healey FP (1982) Phosphate. The biology of Cyanobacteria:105–124
26. Hoang HTT, Duong TT, Nguyen KT, Le QTP, Luu MTN, Trinh DA, Le AH, Ho CT, Dang KD, Némery J, Orange D, Klein J (2018) Impact of anthropogenic activities on water quality and plankton communities in the Day River (Red River Delta, Vietnam). Environ Monit Assess 190(2). doi:10.1007/s10661-017-6435-z
27. Kassambara A (2017) Practical guide to principal component methods in R: PCA, M (CA), FAMD, MFA, HCPC, factoextra, vol 2. Sthda
28. Komárek J, Anagnostidis K (1988) Modern approach to the classification system of cyanophytes. 3 - Oscillatoriales. Algological Studies/Archiv für Hydrobiologie, Supplement Volumes 50–53:327–472
29. Komárek J, Anagnostidis K (1999) Bd. 19/1: Cyanoprokaryota: teil 1: Chroococcales. Fischer, Jena [etc.]
30. Komárek J, Anagnostidis K (2005) Bd. 19/2: Cyanoprokaryota: teil 2: Oscillatoriales. Elsevier, München
31. Krammer K (1991) Suβwasserflora von Mitteleuropa. Bacillariophyceae. 3. Teil: Centrales, Fragilariaceae, Eunotiaceae. Suβwasserflora von Mitteleuropa

32. Lahens L, Strady E, Kieu-Le T-C, Dris R, Boukerma K, Rinnert E, Gasperi J, Tassin B (2018) Macroplastic and microplastic contamination assessment of a tropical river (Saigon River, Vietnam) transversed by a developing megacity. Environ Pollut 236:661–671. doi:10.1016/j.envpol.2018.02.005
33. Lê S, Josse J, Husson F (2008) FactoMineR: AnRPackage for Multivariate Analysis. J Stat Softw 25(1). doi:10.18637/jss.v025.i01
34. Lehman PW, Teh SJ, Boyer GL, Nobriga ML, Bass E, Hogle C (2009) Initial impacts of Microcystis aeruginosa blooms on the aquatic food web in the San Francisco Estuary. Hydrobiologia 637(1):229–248. doi:10.1007/s10750-009-9999-y
35. Machado KB, Vieira LCG, Nabout JC (2018) Predicting the dynamics of taxonomic and functional phytoplankton compositions in different global warming scenarios. Hydrobiologia 830(1):115–134. doi:10.1007/s10750-018-3858-7
36. Marinho MM, Huszar VdM (2002) Nutrient availability and physical conditions as controlling factors of phytoplankton composition and biomass in a tropical reservoir (Southeastern Brazil). Fundamental Applied Limnology 153(3):443–468. doi:10.1127/archiv-hydrobiol/153/2002/443
37. Mhlanga L, Day J, Cronberg G, Chimbari M, Siziba N, Annadotter H (2006) Cyanobacteria and cyanotoxins in the source water from Lake Chivero, Harare, Zimbabwe, and the presence of cyanotoxins in drinking water. African Journal of Aquatic Science 31(2):165–173. doi:10.2989/16085910609503888
38. Morin S, Vivas-Nogues M, Duong TT, Boudou A, Coste M, Delmas F (2007) Dynamics of benthic diatom colonization in a cadmium/zinc-polluted river (Riou-Mort, France). fal 168(2):179–187. doi:10.1127/1863-9135/2007/0168-0179
39. Nguyen NJ, Gratiot N, Garnier J, Strady E, Nguyen DP, Tran VQ, Nguyen AT, Cao ST, Huynh TPT (2020a) Nutrient budgets in the Saigon–Dongnai River basin: Past to future inputs from the developing Ho Chi Minh megacity (Vietnam). River Res Appl 36(6):974–990. doi:10.1002/rra.3552
40. Nguyen NJ, Gratiot N, Strady E, Tran VQ, Nguyen AT, Aimé J, Peyne A (2019a) Nutrient dynamics and eutrophication assessment in the tropical river system of Saigon – Dongnai (southern Vietnam). Sci Total Environ 653:370–383. doi:10.1016/j.scitotenv.2018.10.319
41. Nguyen AT, Némery J, Gratiot N, Garnier J, Dao TS, Thieu V, Laruelle GG (2021) Biogeochemical functioning of an urbanized tropical estuary: Implementing the generic C-GEM (reactive transport) model. Science of the Total Environment:147261. doi:10.1016/j.scitotenv.2021.147261
42. Nguyen H-Q, Ha N-T, Pham T-L (2020b) Inland harmful cyanobacterial bloom prediction in the eutrophic Tri An Reservoir using satellite band ratio and machine learning approaches. Environ Sci Pollut Res 27(9):9135–9151. doi:10.1007/s11356-019-07519-3
43. Nguyen T, Némery J, Gratiot N, Garnier J, Strady E, Tran VQ, Nguyen AT, Nguyen TN, Golliet C, Aimé J (2019b) Phosphorus adsorption/desorption processes in the tropical Saigon River estuary (Southern Vietnam) impacted by a megacity. Estuar Coast Shelf Sci 227(March):106321. doi:10.1016/j.ecss.2019.106321

44. Nguyen TTL, Hoang TH, Nguyen TK, Duong TT (2017) The occurrence of toxic cyanobacterium Cylindrospermopsis raciborskii and its toxin cylindrospermopsin in the Huong River, Thua Thien Hue province, Vietnam Environ Monit Assess 189(10). doi:10.1007/s10661-017-6209-7
45. Oksanen J, Blanchet FG, Friendly M, Kindt R, Legendre P, McGlinn D, Minchin PR, O'Hara RB, Simpson GL, Solymos P (2019) vegan: Community Ecology Package. R package version 2.5–6. 2019
46. Paerl HW, Huisman J (2009) Climate change: a catalyst for global expansion of harmful cyanobacterial blooms. Environmental Microbiology Reports 1(1):27–37. doi:10.1111/j.1758-2229.2008.00004.x
47. Pham T-L, Dao T-S, Tran N-D, Nimptsch J, Wiegand C, Motoo U (2017) Influence of environmental factors on cyanobacterial biomass and microcystin concentration in the Dau Tieng Reservoir, a tropical eutrophic water body in Vietnam. Annales de Limnologie - International Journal of Limnology 53:89–100. doi:10.1051/limn/2016038
48. Reynolds CS (2006) The Ecology of Phytoplankton. Cambridge University Press
49. Rochelle-Newall EJ, Chu VT, Pringault O, Amouroux D, Arfi R, Bettarel Y, Bouvier T, Bouvier C, Got P, Nguyen TMH, Mari X, Navarro P, Duong TN, Cao TTT, Pham TT, Ouillon S, Torréton JP (2011) Phytoplankton distribution and productivity in a highly turbid, tropical coastal system (Bach Dang Estuary, Vietnam). Mar Pollut Bull 62(11):2317–2329. doi:10.1016/j.marpolbul.2011.08.044
50. Sabour B, Loudiki M, Vasconcelos V (2009) Growth responses ofMicrocystis ichthyoblabeKützing andAnabaena aphanizomenoidesForti (cyanobacteria) under different nitrogen and phosphorus conditions. Chem Ecol 25(5):337–344. doi:10.1080/02757540903193130
51. Shapiro J (1990) Current beliefs regarding dominance by blue-greens: The case for the importance of CO2and pH. SIL Proceedings, 1922–2010 24(1):38–54. doi: 10.1080/03680770.1989.11898689
52. Sivonen K (1990) Effects of light, temperature, nitrate, orthophosphate, and bacteria on growth of and hepatotoxin production by Oscillatoria agardhii strains. Appl Environ Microbiol 56(9):2658–2666. doi:10.1128/aem.56.9.2658-2666.1990
53. Smayda TJ, Sournia A (1978) Phytoplankton manual. Phytoplankton to biomass. UNESCO, Paris, pp 273–279
54. Soares MCS, Huszar VLM, Roland F (2007) Phytoplankton dynamics in two tropical rivers with different degrees of human impact (southeast Brazil). River Research Applications 23(7):698–714. doi:10.1002/rra.987
55. Strady E, Dang VBH, Némery J, Guédron S, Dinh QT, Denis H, Nguyen PD (2017) Baseline seasonal investigation of nutrients and trace metals in surface waters and sediments along the Saigon River basin impacted by the megacity of Ho Chi Minh (Vietnam). Environ Sci Pollut Res 24(4):3226–3243. doi:10.1007/s11356-016-7660-7
56. Sugimura Y, Suzuki Y (1988) A high-temperature catalytic oxidation method for the determination of non-volatile dissolved organic carbon in seawater by direct injection of a liquid sample. Mar Chem 24(2):105–131. doi:10.1016/0304-4203(88)90043-6

57. Tran Ngoc TD, Perset M, Strady E, Phan T, Vachaud G, Quertamp F, Gratiot N (eds) (2016) Ho Chi Minh City growing with water-related challenges
58. Triest L, Lung'ayia H, Ndiritu G, Beyene A (2012) Epilithic diatoms as indicators in tropical African rivers (Lake Victoria catchment). Hydrobiologia 695(1):343–360. doi:10.1007/s10750-012-1201-2
59. Trieu NA, Hiramatsu K, Harada M (2014) Optimizing the rule curves of multi-use reservoir operation using a genetic algorithm with a penalty strategy. Paddy Water Environ 12(1):125–137. doi:10.1007/s10333-013-0366-2
60. Twining BS, Baines SB (2013) The trace metal composition of marine phytoplankton. Annual review of marine science 5:191–215. doi:10.1146/annurev-marine-121211-172322
61. van Chu T, Torréton JP, Mari X, Nguyen HMT, Pham KT, Pham TT, Bouvier T, Bettarel Y, Pringault O, Bouvier C, Rochelle-Newall E (2014) Nutrient ratios and the complex structure of phytoplankton communities in a highly turbid estuary of Southeast Asia. Environ Monit Assess 186(12):8555–8572. doi:10.1007/s10661-014-4024-y
62. Varol M, Şen B (2018) Abiotic factors controlling the seasonal and spatial patterns of phytoplankton community in the Tigris River, Turkey. River Res Applic 34(1):13–23. doi:10.1002/rra.3223
63. Wang J, Yuan S, Tang L, Pan X, Pu X, Li R, Shen C (2020) Contribution of heavy metal in driving microbial distribution in a eutrophic river. Sci Total Environ 712:136295. doi:10.1016/j.scitotenv.2019.136295
64. Wehr JD, Descy J-P (1998) USE OF PHYTOPLANKTON IN LARGE RIVER MANAGEMENT. J Phycol 34(5):741–749. doi:10.1046/j.1529-8817.1998.340741.x
65. Wetzel RG (2001) Limnology: lake and river ecosystems. gulf professional publishing
66. Whitton BA (1992) Diversity, Ecology, and Taxonomy of the Cyanobacteria. Photosynthetic Prokaryotes. http://dx.doi.org/10.1007/978-1-4757-1332-9_1
67. Xu C, Kohler TA, Lenton TM, Svenning J-C, Scheffer M (2020) Future of the human climate niche. Proc Natl Acad Sci USA 117(21):11350–11355. doi:10.1073/pnas.1910114117
68. Yang J, Wang F, Lv J, Liu Q, Nan F, Liu X, Xu L, Xie S, Feng J (2020) The spatiotemporal contribution of the phytoplankton community and environmental variables to the carbon sequestration potential in an urban river. Environ Sci Pollut Res 27(5):4814–4829
69. Zhang Y, Prepas EE (1996) Regulation of the dominance of planktonic diatoms and cyanobacteria in four eutrophic hardwater lakes by nutrients, water column stability, and temperature. Can J Fish Aquat Sci 53(3):621–633. doi:10.1139/f95-205
70. Zhao W, Li Y, Jiao Y, Zhou B, Vogt R, Liu H, Ji M, Ma Z, Li A, Zhou B, Xu Y (2017) Spatial and Temporal Variations in Environmental Variables in Relation to Phytoplankton Community Structure in a Eutrophic River-Type Reservoir. Water 9(10):754. doi:10.3390/w9100754

# Figures

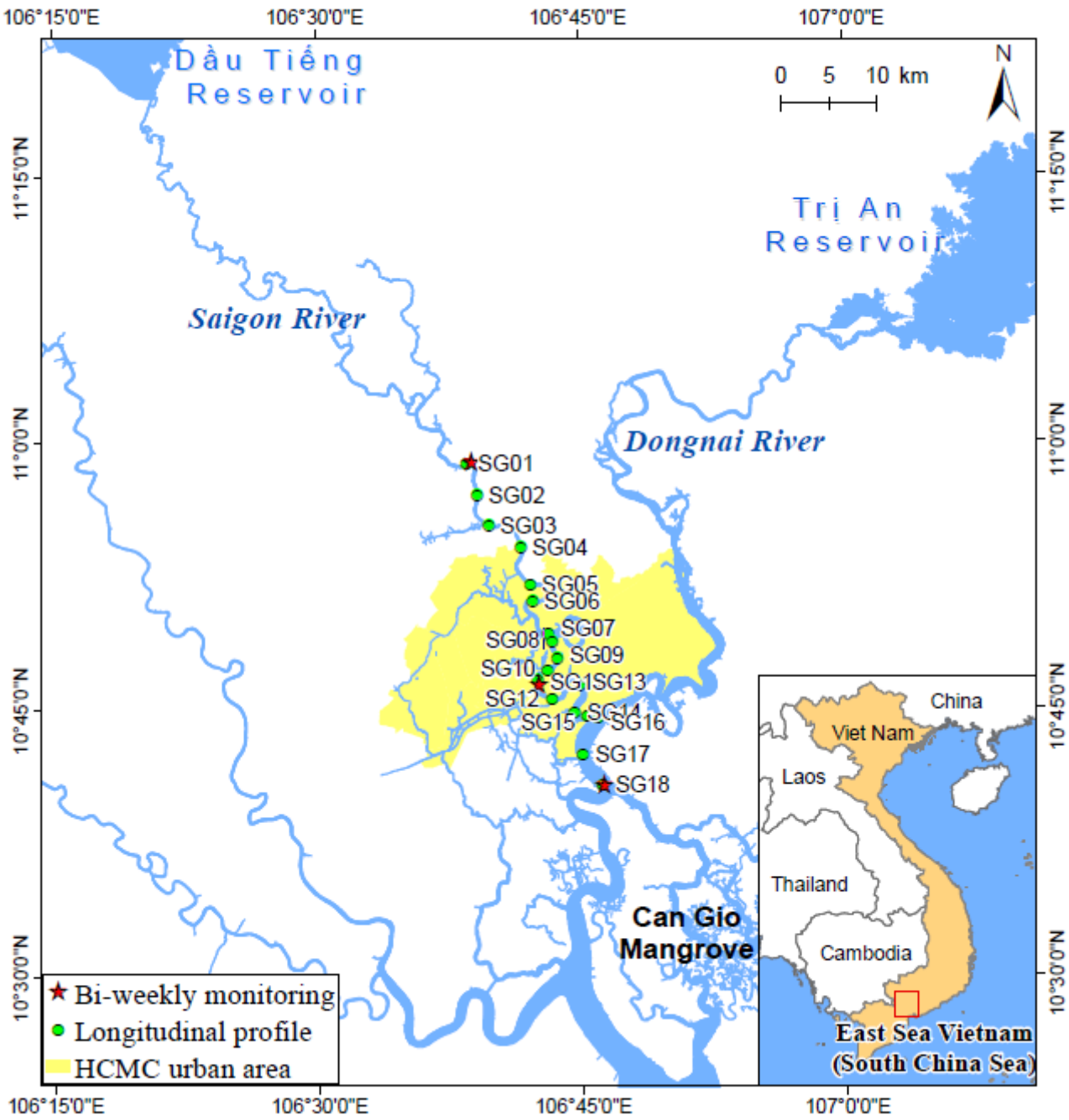


**Figure 1**

Map of Saigon River and location of longitudinal profile and HCMC urban area. Note: The designations employed and the presentation of the material on this map do not imply the expression of any opinion whatsoever on the part of Research Square concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. This map has been provided by the authors.

# Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- Suppl1longitudinalprofiledata.pdf
- Suppl2Significanttest.pdf