Taxonomic and morphofunctional phytoplankton response to environmental variability in rivers from different hydrographic basins in Southern Brazil

Resposta taxonômica e morfofuncional do fitoplâncton à variabilidade ambiental em rios de diferentes bacias hidrográficas no sul do Brasil

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Abstract: Aim: Urbanization, agriculture, and deforestation are the main anthropogenic factors that modify the soil, altering the quality of water, and influencing limnological aspects and the aquatic biota in rivers. We investigated the morphology-based taxonomic and functional response (MBFG) of the phytoplankton community among different public supply rivers in distinct hydrographic basins with ultraoligotrophic, oligotrophic, and mesotrophic characteristics. Methods: We sampled the phytoplankton community and environmental variables in nine rivers along three hydrographic basins in western Paraná. In order to evaluate the taxonomic and functional relationship of the community with the environmental variables, we applied both variance and redundancy analyses. Results: Differences in temperature, pH, turbidity, total phosphorus, chemical oxygen demand, and total dissolved solids were identified among river basins and/or trophic states. The highest taxonomic contributions to richness and biovolume were from green algae and diatoms, while the highest functional contributions were from MBFG IV (algae without specialized traits), MBFG V (unicellular flagellated algae), MBFG VI (algae with a siliceous exoskeleton) and MBFG (large colonial algae). The taxonomic approach was sensitive to environmental variability in the rivers, while for the functional approach no relationship to environmental variability was identified. Conclusions: The taxonomic approach of the phytoplankton community was more sensitive to the environmental variability of the studied rivers than the functional approach based on morphology. Therefore, we reinforce the importance of biological indicators for understanding the dynamics in aquatic ecosystems, providing crucial information for the management of water resources used for public supply.

Keywords: lotic environments; bioindicators; MBFG; water quality.
1. Introduction

Water consumption in Brazil has increased by approximately 80% in the last two decades and is projected to increase by 23% until 2030, with urban water supply being the second largest use of Brazilian water bodies (ANA, 2020). Urban supply is lower only when compared to use in agriculture, which represents a service of exceptional economic value, and due to the intensification of agricultural activities, competition has been created between supply and regulation services in productive landscapes (Jia et al., 2014). However, the high demand for water resources, whether for supply or for agriculture, stemming from population growth and economic development, drastically affects the sustainability of aquatic ecosystems (Gleick, 2018). Therefore, assessing the environmental quality of rivers is of utmost importance for sustainable hydrographic basin management since the anthropic influence and exploitation of the surrounding landscape affects water quality and its multiple uses (Kashaigili, 2008; Santos et al., 2018).

Currently, the main landscape-modifying anthropic activities are related to urbanization, agriculture, and deforestation (Vörösmarty et al., 2010; Yu et al., 2013; Kim et al., 2019), directly influencing ecosystems at local, regional, and global scales (Wan et al., 2014). These activities generate impacts on the limnological characteristics of rivers, increasing the concentration of nutrients and pollutants (fertilizers, pesticides, and sewage flows), and altering water quality (Schulz & Martins-Junior, 2001; Bussi et al., 2016; Xiao et al., 2019; Zhang et al., 2019), and, consequently, aquatic biota (Medeiros et al., 2020). Thus, in addition to the assessment of physical and chemical water characteristics, complementary methods using biological indicators (Soofiani et al., 2012; Wang et al., 2021) are efficient strategies for understanding how anthropic activities and hydrographic basin uses influence river water quality and are important for establishing monitoring strategies.

There are still few studies that evaluate the phytoplankton of rivers and its relationship to environmental dynamics when compared to lakes and reservoirs (Bolgovics et al., 2017), especially in Brazil’s extensive hydrographic network. The ecological knowledge of phytoplankton dynamics in rivers has intensified since the first half of the 20th century. From the studies of Colin Reynolds (2006), who identified that river discharge and turbidity were the main barriers to the development and maintenance of phytoplankton in unidirectional flow, more studies have been developed (Abonyi et al., 2021). However, due to the intense global changes that aquatic ecosystems have been facing, additional studies should be developed in order to establish a relationship between phytoplankton communities and environmental variability (Bolgovics et al., 2017).

The phytoplankton community is an important component of aquatic ecosystems, contributing significantly to primary productivity, and being a key link in nutrient cycling (Litchman et al., 2015). Algae and cyanobacteria from this group...
are commonly used as bioindicators to assess anthropogenic impacts in freshwater environments (Kim et al., 2019). These organisms reflect different responses (e.g., physiological responses, changes in abundance, changes in community structure and productivity) depending on the intensity of stressors and anthropogenic changes (Salmaso & Tolotti, 2021). Phytoplankton responds quickly and efficiently to changes in water conditions, such as levels of nutrients and toxic contaminants, electrical conductivity, turbidity, and pH (Triest et al., 2012; Castro-Roa & Pinilla-Agudelo, 2014; Jia et al., 2019). Their short life cycle, representative population size, easy sampling, and storage (Litchman & Klausmeier, 2008; Kruk et al., 2017) also facilitate their use in monitoring programs. Changes in the occurrence and distribution of taxa, as well as in their population size, morphology, and physiology, are some of the main responses as a function of anthropogenic activities on the ecosystem (Casé et al., 2008; Abonyi et al., 2020; Zohary et al., 2021). Thus, phytoplankton species composition, richness, and abundance represent important measures for assessing the health and water quality of rivers (Soofiani et al., 2012; Santana et al., 2016; Zhang et al., 2020), as phytoplankton directly reflect ecosystem functioning (Borics et al., 2021). The use of functional groups has also been applied and shown to be efficient in water quality assessment (Salmaso et al., 2015; Graco-Roza et al., 2021).

Therefore, in order to understand the structure and functionality of phytoplankton and the patterns of their dynamics in aquatic environments, the phytoplankton community can be grouped based on morphological similarity of species (e.g., volume, size, presence or absence of flagella) (Kruk et al., 2010; Kruk & Segura, 2012). In addition to taxonomic characteristics and traditional measures of community structure, morphology-based functional measures provide crucial information about environmental variability in aquatic ecosystems, being relatively easy to measure.

In view of the need to evaluate the water quality of catchment rivers and their use for public supply, this study aims to evaluate the phytoplankton community based on the taxonomic and morphofunctional approaches as models of response to the environmental variability of rivers in southern Brazil. Thus, we investigate the water quality of nine supply rivers with surrounding urbanization and agricultural characteristics and relate it to phytoplankton distribution. Therefore, we have as central questions of our study: i) how does the taxonomic and morphofunctional distribution of phytoplankton relate to the environmental conditions of the supply rivers, considering their different trophic states and different hydrographic basins, and ii) how this relationship can indicate the environmental quality of these rivers.

2. Materials and Methods

2.1. Selection, location, and characterization of the study sites

The state of Paraná has 16 hydrographic basins (Resolution n. 024/2006/SEMA) (Paraná, 2006): Litorânea, Iguaçu, Ribeira, Itararé, Cinzas, Tibagi, Ivaí, Paranapanema I, II, III and IV, Pirapó, Paraná I, II and III, and Piquiri. The western region of the state, which covers 54 municipalities, is bordered by the Paraná III, Piquiri, and Iguaçu basins (AMOP, 2018). This region has an economy focused on agricultural activities, and a large part of the municipalities are dominated by extensive monoculture areas and urbanized areas (PNUD, 2018), including the most populous municipalities, such as Cascavel (336,073 inhabitants), Foz do Iguaçu (257,971 inhabit.), Toledo (144,601 inhab.), and Medianeira (46,940 inhab.) (IBGE, 2010).

We selected nine rivers used for water withdrawal for public supply in the western region of Paraná, along the hydrographic basins of the Lower Iguaçu River (BI), Paraná III (PILL), and Piquiri (PQ), which are distributed in nine municipalities: Guaraíçu (GUAR), Catanduvas (CTD), Três Barras do Paraná (TBP), Boa Vista Aparecida (BVA), Foz do Iguaçu (FOZ), Medianeira (MED), Santa Tereza do Oeste (STO), Cascavel (VEL), and Toledo (TOL) (Figure 1). Information about the sampling sites, main anthropic activities in the region, trophic state, morphometric and hydrological characteristics are presented in Table 1.

The water samples for physicochemical and biological analysis were taken in two locations in each river, one at the headwaters and another at the point of the catchment used for water supply, during the summer of 2020, totaling 18 samples. All samples were deposited in the herbarium of UNIOESTE - Universidade Estadual do Oeste do Paraná - UNOPA, Cascavel campus, linked to the Brazilian Herbaria Network, and the data were computerized and made available on Species Link (2021).

2.2. Sampling and analysis of environmental variables in rivers

Precipitation data (Pre) were provided by the Paraná Meteorological Institute (Simepar). Data...
on maximum depth ($Z_{\text{max}}$ - cm), water temperature (Temp - °C), dissolved oxygen (DO - mg L$^{-1}$), pH, electrical conductivity (Conduct - mS cm$^{-1}$), and turbidity (Turb - NTU) were measured at the time of sampling using a Horiba U-5000 multiparameter probe. Data regarding flow ($m^3 s^{-1}$) and maximum depth were collected in situ using a ruler, measuring tape, and a floating object. The flow was calculated by multiplying the average speed resulting from the displacement of the object and the cross-sectional area where the stone was collected, measured in situ (Table 1).

Chemical analyses were measured based on water samples collected by sub-surface immersion of polyethylene bottles, properly cooled, and kept in the dark until their destination. The oxygen consumption that occurred due to chemical oxidation was evaluated through the chemical oxygen demand (COD - mg L$^{-1}$) and organic matter was evaluated through the biochemical oxygen demand (BOD - mg L$^{-1}$) and was estimated following the methods described in Standard Methods (APHA, 2017). Concentrations of nitrate (NO$_3^-$ - mg L$^{-1}$), ammoniacal nitrogen (N-NH$_3$ - mg L$^{-1}$), total phosphorus (TP - mg L$^{-1}$), orthophosphate (PO$_4^{3-}$ - mg L$^{-1}$), chlorophyll $a$ (CL$a$ - mg L$^{-1}$), and total dissolved solids (TDS - mg L$^{-1}$) were also estimated (APHA, 2017).

2.3. TSI – Trophic State Index

The Trophic State Index presented and used in the calculation of the Aquatic Life Protection Index (ALPI), was composed of the Trophic State Index for phosphorus - TSI(PT) and the Trophic State Index for chlorophyll $a$ - TSI(CL), modified by Lamparelli (2004), being established for lotic environments, according to the Equations 1 and 2:

\[
\text{TSI(PT)} = 10^6 \left( -0.42 - 0.36 \ln(\text{TP}) / \ln(2) \right)^{20}
\]

\[
\text{TSI(CL)} = 10^6 \left( -0.7 - 0.6 \ln(\text{CT}) / \ln(2) \right)^{20}
\]

where:

- TP: total phosphorus concentration measured at the water surface, in µg L$^{-1}$;
- CL: chlorophyll $a$ concentration measured at the water surface, in µg L$^{-1}$; ln: natural logarithm.

For the classification of Trophic State for rivers the Carlson Index (Carlson, 1977) modified by Toledo et al. (1983) was used.

2.4. Sampling and analysis of the phytoplankton community

The qualitative analyses were based on phytoplankton samples collected using a plankton net of ~20 µm mesh size and preserved in Transeau solution (Bicudo & Menezes, 2017), in order to concentrate the phytoplankton material and facilitate taxonomic studies. The qualitative study of phytoplankton was performed in an Olympus CX41 photomicroscope, with an Olympus SC30 camera attached, and the morphometry of the taxa was performed at 400× and 1000× magnification. We followed the classification system of Round (1965, 1971) proposed by Bicudo & Menezes (2017), to group the algae at the Class level.

For the quantitative analyses, the phytoplankton community was collected directly with 300 mL flasks at the subsurface and fixed in situ with acetic Lugol. The quantitative analysis was estimated according to the methodology described by Utermöhl (1958), using an Olympus inverted microscope, model CKX41. The sedimentation volume was defined according to the concentration of algae and/or detritus present in the sample, with the sedimentation time being equivalent.
Table 1. Geographic location of the sampled rivers in different hydrographic basins in southern Brazil (1 - headwaters; 2 - water catchment point), main anthropic activities, trophic level according to Lamparelli (2004), and morphometric and hydrological characteristics.

| Hydrographic basins | Municipality | River | Sampling sites | Latitude Longitude | Anthropic activity | Trophic state | Depth (cm) | Width (m) | Flow (m³/s) |
|---------------------|--------------|-------|----------------|--------------------|--------------------|---------------|------------|-----------|-------------|
| Piquiri             | Guaraniçu    | Baú River | GUARP1 | 25°40'56"S 52°53'29"W | Agricultural production PP | Mesotrophic | 14 | 5.78 | 0.05 |
|                     |              |       | GUARP2 | 25°40'27"S 52°53'20"W | Agricultural production PP | Ultraoligotrophic | 18 | 4.69 | 0.11 |
| Paraná III          | Medianeira   | Alegria River | MEDP1 | 25°18'35"S 54°30'31"W | Agricultural production PP | Urban | 8 | 2.90 | 0.04 |
| Toledo              | Toledo River | TOLP1 | 25°17'30"S 54°40'35"W | Agricultural production GP | Urban | 34 | 3.70 | 0.09 |
|                     |              |       | TOLP2 | 53°39'50"W 53°42'40"W | Agricultural production GP | Urban | 52 | 7.90 | 3.00 |
| Lower Iguaçu River  | Cascavel     | Cascavel River | CVELP1 | 52°53'29"S 52°26'06"W | Agricultural production GP | Ultraoligotrophic | 30 | 4.80 | 0.28 |
|                     |              |       | CVELP2 | 52°53'20"S 53°26'19"W | Agricultural production GP | Ultraoligotrophic | 18 | 9.50 | 0.05 |
|                     | Arroio Passo | CTDP1 | 25°11'13"S 53°08'18"W | Agricultural production PP | Urban | 17 | 7.25 | 0.30 |
|                     | Liso River   | CTDP2 | 25°12'38"S 53°07'51"W | Agricultural production PP | Urban | 24 | 6.90 | 0.27 |
|                     | Jacutinga River | BVAP1 | 25°25'17"S 53°25'46"W | Agricultural production GP | Urban | 28 | 3.50 | 0.18 |
|                     | Tamanduá River | FOZP1 | 25°30'26"S 54°31'50"W | Agricultural production GP | Ultraoligotrophic | 9 | 2.86 | 0.06 |
|                     |              |       | FOZP2 | 52°32'13"S 54°31'25"W | Agricultural production GP | Urban | 30 | 3.40 | 0.15 |
|                     | Gonçalves Dias River | STOP1 | 25°20'29"S 53°35'20"W | Agricultural production GP | Ultraoligotrophic | 19 | 1.50 | 0.16 |
|                     |                | STOP2 | 25°30'47"S 53°36'14"W | Agricultural production GP | Urban | 21 | 5.00 | 0.23 |
|                     | Itaguaçu stream | TBPP1 | 25°26'11"S 53°11'17"W | Agricultural production GP | Oligotrophic | 40 | 2.40 | 0.06 |
|                     |                | TBPP2 | 25°26'21"S 53°10'50"W | Agricultural production GP | Oligotrophic | 25 | 3.20 | 0.15 |

PP = small-scale production; GP = large-scale production.
to the height of the counting chamber used (Margalef, 1983). Counting was performed in random transects, and the counting limit was established by the species rarefaction curve, basing the counting of individuals on the form they occur in nature: cells, colonies, coenobium, or filaments (Uhelinger, 1964). Phytoplankton density was calculated according to APHA (2017), and results were expressed as individuals per milliliter (ind. mL\(^{-1}\)). The phytoplankton biovolume was calculated by multiplying the density of each taxon by its respective volume. The cell volume was calculated from geometric models, according to the shape of the cells (Sun & Liu, 2003). Total phytoplankton richness was defined by the number of taxa found in each quantitative sample.

We classified the phytoplankton taxa recorded in the quantitative samples into morphology-based functional groups (MBFG), according to Kruk et al. (2010) and Reynolds et al. (2014). This classification uses continuous (individual volume, surface area, maximum linear dimension, mean, and range) and categorical (presence and frequency of aerotopes, flagella, mucilage, heterocysts, silica exoskeletal structures, and mean frequency) variables for grouping taxa into MBFGs. Thus, according to these criteria, phytoplankton organisms were classified into MBFG I - small organisms with high individual volume to surface area ratio. Species in this group are small-sized with rapid individual growth rate and high numerical abundance; MBFG II - small flagellate organisms with silica exoskeletal structures. They have low biomass and do not pose a significant threat to water quality; MBFG III - species with large filaments and aerotopes. They are large and slow-growing organisms, but their high volume/surface area ratio gives them greater tolerance to light limiting conditions; MBFG IV - medium-sized organisms with no specialized traits and a moderate tolerance for resource limitation; MBFG V - medium to large unicellular flagellates. The medium size and volume to surface area ratio, the presence of flagella, and the production of cysts allow these organisms to tolerate low levels of nutrients; MBFG VI - non-flagellated organisms with silica exoskeletal. Represented only by diatoms, containing high cell density; MBFG VII - large mucilaginous colonies; MBFG VIII - nitrogen-fixing cyanobacteria. The high size and volume and the low volume/surface area ratio tend to make the species of this group sensitive to low resource supply. They can produce toxins and allelopathic substances.

2.5. Data analysis strategies

We compared possible statistical differences of environmental variables among the trophic states of the rivers and hydrographic basins evaluated, applying an analysis of variance (two-way ANOVA). In addition, the environmental variables were submitted to Principal Component Analysis (PCA), aiming to characterize and order the sampling sites according to environmental variability. For the PCA the following variables were used: water temperature, dissolved oxygen, pH, electrical conductivity, turbidity, maximum depth, total phosphorus, orthophosphate, nitrate, ammonial nitrogen, chemical oxygen demand, biochemical oxygen demand, and total dissolved solids.

Differences in biovolume and richness of taxa and MBFGs among hydrographic basins, as well as river trophic status, were assessed using a non-parametric permutational analysis of variance (PERMANOVA; Anderson, 2001). Subsequently, to evaluate the biovolume relationship of the phytoplankton community (taxonomic and morphofunctional matrix) with environmental variables, a Redundancy Analysis (RDA) was performed. The community biovolume data underwent Hellinger’s transformation (Borcard et al., 2011). The collinearity of the environmental variables (water temperature, dissolved oxygen, pH, electrical conductivity, turbidity, maximum depth, total phosphorus, orthophosphate, nitrate, ammonial nitrogen, chemical oxygen demand, biochemical oxygen demand and total dissolved solids) was tested using the VIF (Variance Inflation Factor - VIF > 10 were removed – Oksanen et al., 2017) and then the selection procedure (Ordistep) was applied (p<0.05) (Rao, 1964). All analyses were performed using the R language and environment for computational statistics (R Development Core Team, 2014), with the Vegan package (Oksanen et al., 2017).

3. Results

3.1. Environmental characterization of rivers

We observed differences between the mean values of temperature, pH, COD, and TDS among the hydrographic basins, while differences between pH, turbidity, and TP were identified among the trophic states of the rivers (Table 2). Mean values and standard deviation of environmental variables are also presented in Table 2.

The principal component analysis (PCA) summarized for the first axis 39%, and for the
Table 2. Mean, standard deviation, and ANOVA results (among hydrographic basins and river trophic status) applied to the environmental variables sampled in different supply rivers in the southern region of Brazil.

| Municipalities Basins | BVA Lower Iguassu River | CTD Lower Iguassu River | CVEL Lower Iguassu River | FOZ Lower Iguassu River | GUAR Piquiri III | STO Lower Iguassu River | TBP Lower Iguassu River | STO Paraná III | Basins | Trophy | Basins x Trophy |
|-----------------------|--------------------------|-------------------------|--------------------------|-------------------------|-----------------|------------------------|------------------------|-----------------|--------|--------|-----------------|
| **Environmental variables** |                          |                          |                          |                          |                 |                        |                        |                 |        |        |                 |
| Temp (ºC)              | 23.22 ± 1.08             | 19.45 ± 0.26            | 20.9 ± 1.00              | 22.41 ± 0.98            | 17.93 ± 0.42    | 22.52 ± 1.26         | 21.04 ± 0.05          | 20.83 ± 0.20     | 0.018  | 0.45   | 0.94            |
| DO (mg L⁻¹)            | 3.43 ± 0.32              | 4.94 ± 0.35             | 10.74 ± 0.35            | 2.58 ± 0.54             | 4.84 ± 0.22     | 2.8 ± 0.57            | 2.53 ± 0.78           | 4.47 ± 0.35        | 10.59  | 0.64   | 0.05            |
| pH                    | 7.41 ± 0.28              | 7.31 ± 0.01             | 5.47 ± 0.52             | 6.32 ± 0.13             | 6.79 ± 0.44     | 6.66 ± 0.10           | 6.64 ± 0.31           | 7.27 ± 0.09        | 5.94   | 0.04   | 0.12            |
| Conduct (mS cm⁻¹)      | 0.08 ± 0.01              | 0.08 ± 0.01             | 0.05 ± 0.01             | 0.05 ± 0.01             | 0.08 ± 0.01     | 0.03 ± 0.01           | 0.01 ± 0.01           | 0.08 ± 0.01        | 0.02   | 0.02   | 0.03            |
| Z_max (m)              | 0.26 ± 0.02              | 0.17 ± 0.01             | 0.41 ± 0.15             | 0.19 ± 0.15             | 0.16 ± 0.03     | 0.21 ± 0.18           | 0.2 ± 0.01            | 0.32 ± 0.10        | 0.44   | 0.03   | 0.48            |
| Turb (NTU)             | 9.21 ± 0.46              | 6.07 ± 2.31             | 9.67 ± 2.31             | 13.65 ± 3.06            | 13.25 ± 1.20    | 8.91 ± 1.20           | 10.53 ± 1.51          | 9.61 ± 0.21        | 22.25  | 0.099  | 0.028           |
| BOD (mg L⁻¹)           | 1.30 ± 0.36              | 1.00 ± 0.01             | 1.68 ± 0.82             | 1.75 ± 1.06             | 1.76 ± 0.14     | 1.05 ± 0.08           | 1.45 ± 0.64           | 1.00 ± 0.01        | 1.72   | 0.31   | 0.07            |
| COD (mg L⁻¹)           | 67.50 ± 3.53             | 58.33 ± 2.35            | 41.66 ± 9.42            | 16.66 ± 11.78           | 63.33 ± 14.14   | 61.67 ± 14.14         | 71.66 ± 2.35          | 77.50 ± 5.89       | 25.83  | 0.04   | 0.25            |
| TP (mg L⁻¹)            | 0.02 ± 0.01              | 0.02 ± 0.02             | 0.02 ± 0.02             | 0.02 ± 0.05             | 0.02 ± 0.02     | 0.02 ± 0.02           | 0.01 ± 0.01           | 0.03 ± 0.01        | 0.04   | 0.03   | 0.34            |
| NO₃ (mg L⁻¹)           | 0.09 ± 0.07              | 1.00 ± 0.02             | 1.25 ± 0.49             | 1.25 ± 0.40             | 0.80 ± 0.42     | 1.00 ± 0.01           | 0.55 ± 0.07           | 1.75 ± 0.49        | 1.80   | 0.32   | 0.47            |
| N-NH₃ (mg L⁻¹)         | 0.00 ± 0.01              | 0.00 ± 0.01             | 0.05 ± 0.07             | 0.17 ± 0.25             | 0.01 ± 0.01     | 0.01 ± 0.01           | 0.02 ± 0.01           | 0.01 ± 0.01        | 0.01   | 0.67   | 0.37            |
| PO₄ (mg L⁻¹)           | 0.11 ± 0.01              | 0.01 ± 0.02             | 0.02 ± 0.02             | 0.02 ± 0.02             | 0.01 ± 0.01     | 0.01 ± 0.01           | 0.01 ± 0.01           | 0.01 ± 0.01        | 0.06   | 0.06   | 0.59            |
| TDS (mg L⁻¹)           | 77.25 ± 6.72             | 71.00 ± 4.95            | 40.00 ± 3.53            | 41.50 ± 6.36            | 41.50 ± 12.02   | 48.75 ± 5.30          | 40.50 ± 8.48          | 71.75 ± 10.96      | 21.57  | 0.003  | 0.13            |
| Precipitation (mm)     | 5.02 ± 0.68              | 3.81 ± 0.38             | 3.81 ± 2.87             | 2.87 ± 2.6              | 2.54 ± 3.81     | 5.02 ± 2.84           | 2.84 ± 1.68           | -                | -      | -      | -                |
| Flow (m³ s⁻¹)          | 0.22 ± 0.68              | 0.17 ± 0.17             | 0.16 ± 0.10             | 0.18 ± 0.04             | 0.08 ± 0.04     | 0.06 ± 0.03           | 0.19 ± 0.04           | 0.1 ± 0.06         | 1.81   | 1.68   | -                |

Significant differences (p < 0.05) are highlighted in bold. Temp: water temperature, DO: dissolved oxygen, pH: Conduct: electrical conductivity, Z_max: maximum depth, Turb: turbidity, BOD: biochemical oxygen demand, COD: chemical oxygen demand, TP: total phosphorus, NO₃: nitrate, N-NH₃: ammoniacal nitrogen, PO₄: orthophosphate, and TDS: total solids. BVA: Boa Vista da Aparecida, CTD: Catanduvas, CVEL: Cascavel, FOZ: Foz do Iguaçu, GUAR: Guaraniaçu, MED: Medianeira, STO: Santa Tereza do Oeste, TBP: Três Barras do Paraná, and TOL: Toledo.
second axis 16% of the environmental variability among the rivers of the different hydrographic basins (Figure 2). The spread of the scores of the sampled sites in these axes evidenced a separation in the diagram of the sampled basins, especially some rivers located in the Lower Iguassu River and Paraná III basins (Figure 2). The first axis of the PCA explained positively the environmental variability mainly in relation to turbidity (0.34) and negatively with pH (-0.38), TDS (-0.39), COD (-0.34) and \( \text{PO}_4^- \) (-0.31). The second axis was positively related to the variable N-NH\(_3\) (0.53), isolating one of the rivers of the Lower Iguassu River basin, and negatively with Conduct (-0.51) and NO\(_3^-\) (-0.48).

3.2. Phytoplankton community characterization and distribution in rivers

A total of 67 taxa were recorded along all rivers, distributed in eight taxonomic classes: Dinophyceae (1 taxon), Cyanophyceae (1 taxon), Coscinodiscophyceae (3 taxa), Trebouxiophyceae (3 taxa), Zygnematophyceae (8 taxa), Euglenophyceae (8 taxa), Bacillariophyceae (10 taxa) and Chlorophyceae (33 taxa). Desmodesmus (R.Chodat) S.S.An, T.Friedl & E.Hegewald and Ankistrodesmus Corda were the most representative genera, with 11 and 7 taxa, respectively.

Considering taxa richness, the highest values were recorded in the rivers of the Lower Iguassu River and Paraná III basins (Figure 3a). Regarding phytoplankton biovolume, low values were recorded in most rivers, however, three rivers from the Lower Iguassu River basin presented higher values (>1 mm\(^3\) L\(^{-1}\)) (Figure 3b). Green algae, diatoms, and euglenophyceans were the taxonomic groups with the greatest contribution in terms of richness and biovolume.

When we classified the species according to morphological characteristics, we recorded four MBFG: MBFG IV, MBFG V, MBFG VI, and MBFG VII. Most rivers were represented by the presence of MBFG IV, V, and VI (Figure 4a).
Regarding the biovolume of MBFG, the rivers from the Lower Iguassu River basin also showed the highest values, especially due to the presence of MBFG IV (Figure 4b).

According to PERMANOVA, differences were only seen for species richness between hydrographic basins (\(F = 0.19865; p = 0.030\)). For species biovolume and MBFG richness and biovolume, no differences were verified between the hydrographic basins or between the trophic state of the rivers (Table 3).

3.3. Phytoplankton community and environmental variability of rivers

The Redundancy Analysis (RDA) performed with the species biovolume as a function of environmental variables resulted in explanatory power of 25%, also evidencing that the species matrix is significantly related to the selected variables (\(F = 1.63, p = 0.007\); Figure 5). Only the RDA1 axis (RDA1: \(F = 24.513, p = 0.019\); RDA2: \(F = 15.833, p = 0.203\)) was significant. The variables selected as explanatory in the model were turbidity, orthophosphate and nitrate. However, \(\text{NO}_3^-\) (\(F = 21.511; p = 0.004\)) and \(\text{PO}_4^{3-}\) (\(F = 17.124; p = 0.039\)) were significant. It was possible to observe a clear spatial separation between the basins, especially of the Piquiri basin, while the Lower Iguazu River and Paraná III basins were divided into three groups separating the rivers. RDA did not identify any variable explaining the presence of MBFG in the rivers of the different hydrographic basins.

![Figure 4. (a) MBFG richness and (b) MBFG biovolume (mm$^3$ L$^{-1}$) in rivers from different river hydrographic basins in western Paraná, Brazil (1- headwaters; 2 – water catchment point; O – oligotrophic; M – mesotrophic; U – ultraoligotrophic).](image)

Table 3. PERMANOVA results applied to richness and biovolume of taxa and MBFG data to evaluate differences between hydrographic basins and river trophic states (Significance level = \(p < 0.05\), highlighted in bold).

| Hydrographic basin | Taxa       | DF | F          | p        |
|--------------------|------------|----|------------|----------|
| MBFG               | Richness   | 2  | 0.19865    | 0.030    |
|                    | Biovolume  | 2  | 0.15331    | 0.114    |
| MBFG               | Richness   | 2  | 0.18809    | 0.306    |
|                    | Biovolume  | 2  | 0.12170    | 0.487    |
| Trophic state      | Taxa       | DF | F          | p        |
| MBFG               | Richness   | 2  | 0.08837    | 0.752    |
|                    | Biovolume  | 2  | 0.08023    | 0.931    |
| MBFG               | Richness   | 2  | 0.02975    | 0.751    |
|                    | Biovolume  | 2  | 0.03927    | 0.984    |

DF: degree of freedom.
4. Discussion

Our results pointed out a distinction between the rivers of the different hydrographic basins as a function of the environmental variables that characterized the water quality, which is reflected in the phytoplankton community. Urban supply rivers are sensitive to anthropogenic activities, such as the continuous growth of urban populations and agricultural activities, which affect surface waters (Kim et al., 2019; Silva et al., 2020). Moreover, these environments play an important role in maintaining biodiversity and sustaining ecosystem products and services that are also essential for human well-being (Zhang et al., 2019). The trophic state of the different supply rivers evaluated and the main environmental conditions were reflected in the biological response of phytoplankton, which proved to be a good indicator of environmental quality, especially when treated at the taxonomic level.

Relating the community to the environmental conditions of the rivers, the taxonomic approach proved to be more sensitive than the MBFGs. MBFGs have been applied in different studies, and their responses to environmental variability have also been satisfactory (Bohnenberger et al., 2018; Cupertino et al., 2019; Yang et al., 2020; Trindade et al., 2021). However, in our study, MBFGs were not related to the environmental variability of rivers. This is likely associated with the low phytoplankton biovolume in most of the rivers evaluated, as well as the simple and objective refinement of the MBFGs, allowing to fit the algae and cyanobacteria into few groupings, which consequently did not relate to any environmental filter. Thus, our results suggest that the small number of MBFG was not sensitive to environmental variability in rivers, captured by the taxonomic approach alone (Cupertino et al., 2019). The response of MBFGs in lotic environments and their weak relationship to environmental variability has already been reported (Bortolini et al., 2014), and thus the taxonomic approach may still be more sensitive for river biomonitoring when treated on a local scale and with small numbers of samplings.

We identified in this study a higher contribution of green algae, diatoms, and euglenophyceans to richness and biovolume along the rivers. These algae show high sensitivity to environmental changes and are used as bioindicators of organic pollution and water eutrophication in most aquatic ecosystems (Lobo et al., 2016; Barnard et al., 2021).

Green algae belonging to Chlorophyceae presented the highest contribution in biovolume and richness in most of the sampled rivers, which has already been recorded in other studies (Rodrigues et al., 2007; Descy et al., 2011; Zhao et al., 2012; Yang et al., 2019). These algae are considered opportunistic since some species contain small size and fast growth, besides being found in various environments, from oligotrophic waters to polluted environments (Bicudo & Menezes, 2017). They are also resistant to environmental variations and competition with other species (Pan et al., 2011; Domingues & Torgan, 2012). Additionally, several species have accessory structures such as spines or mucilage, helping them to avoid longitudinal carriage and favoring their permanence longer in the water column (Kruk & Segura, 2012). In our study, this group was correlated with ultraoligo- to mesotrophic environments, from the Paraná III and Lower Iguaçu River basins, as well as higher concentrations of NO$\text{}_\text{3}^- $ and PO$\text{}_\text{4}^- $. High levels of these variables in rivers usually occur due to surface runoff and wastewater discharges, influencing water quality, and promoting species selection (Kelly et al., 2015; Shi et al., 2017).

Diatoms in turn have genera related to urban growth associated with organic pollution conditions, containing indicator species of environmental impacts (Moresco & Rodrigues, 2014; Rangel et al., 2017). These aspects are best visualized through ecological guilds for the group. Passy (2007)
classified larger species, such as those in the genus *Pinnularia*, as high profile, commonly found in the phytoplankton and poorly resistant to turbulence, but favored by nutrient enrichment. The genus *Navicula* is included in the motile guild and is considered an adapted genus to turbulence and variations in nutrient concentrations (Passy, 2007). Previous studies corroborate our results, demonstrating that diatoms are representative contributors to taxonomic richness and abundance in the phytoplankton community (Bolgovics et al., 2017; Conceição et al. 2021) of ultraoligotrophic, to mesotrophic, and agricultural or urban rivers, with low flow and distinct nutrient concentrations (Passy, 2007; Simić et al., 2015).

Euglenophyceae have high metabolic flexibility and, exploiting diverse organic carbon sources under different conditions, (Cordoba et al., 2021). This adaptation confers an advantage in oligotrophic or organic matter-rich water bodies, strongly light-limited (Bicudo & Menezes, 2017). The representativeness of this group in our study may be related to their flagellar motility, which facilitates survival in different environmental conditions, such as shallow waters and with little turbulence (Brasil & Huszar, 2011).

The variation in species biovolume among river trophic states reflects the environmental variability among aquatic environments. Thus, the abundance of taxa was associated with environmental conditions such as turbidity, temperature, pH, and nutrient concentrations. Temperature is considered one of the main environmental factors as it has major impacts on phytoplankton growth, leading to seasonal variation in algal abundance and composition (Fietz et al., 2005; Lv et al., 2020). Water quality can also be altered by nutrient concentrations and turbidity, promoting the selection of a single species or a few dominant species that can survive in this stressful environment (Yang et al., 2021). Thus, changes in physical and chemical water conditions indicate impacts from increasing urbanization as well as agricultural activities (Lötjönen & Ollikainen, 2019), regional characteristics adjacent to the rivers evaluated here. Such conditions comprise major ecological axes that determine the niche of species and influence reproduction, resource acquisition, and predation prevention (Litchman & Klausmeier, 2008), reflecting in higher contributions of green algae, diatoms, and euglenophyceans in the phytoplankton community.

We evaluated the limnological characteristics of rivers in different hydrographic basins distributed in nine municipalities of western Paraná, based on physical, chemical, and biological parameters, verifying distinct environmental characteristics among the studied environments. In our study, the taxonomic approach used for the phytoplankton community was more sensitive in responding to the environmental variability of the studied rivers than the MBFG. In general, the biovolume of the classes grouped the environments, with Bacillariophyceae related to ultraoligotrophic environments of the Piquiri and Paraná III basins and mesotrophic environments of the Piquiri and Lower Iguassu River basins, Chlorophyceae related to ultraoligotrophic environments of the Paraná III basin and mesotrophic environments of the Lower Iguassu River, and Euglenophyceae related to oligotrophic environments of the three basins evaluated. Our data indicate environments with intermediate stages of degradation, but facing effects of urbanization and agricultural expansion. The integrity of these rivers is gradually being affected, reflected in the phytoplankton community, showing that these activities can compromise water quality used for public supply. Thus, comprehending the environmental variability and the response of biological indicators is essential to understanding the functions of aquatic ecosystems, as well as the provision of ecosystem services.

**References**

Abonyi, A., Kiss, K.T., Hidas, A., Borics, G., Várbiró, G., & Ács, E., 2020. Cell size decrease and altered size structure of phytoplankton constrain ecosystem functioning in the middle Danube river over multiple decades. Ecosystems 23(6), 1254-1264. PMid:33005096. http://dx.doi.org/10.1007/s10021-019-00467-6.

Abonyi, A., Descy, J.P., Borics, G., & Smeti, E., 2021. From historical backgrounds towards the functional classification of river phytoplankton sensu Colin S. Reynolds: what future merits the approach may hold? Hydrobiologia 848(1), 131-142. http://dx.doi.org/10.1007/s10750-020-04300-3.

Agência Nacional de Águas – ANA, 2020. Conjuntura dos recursos hídricos no Brasil 2020: informe anual. Brasília: ANA, 108 p.

American Public Health Association – APHA, 2017. Standard Methods for the Examination of Wastewater, Washington. DC: APHA.

Anderson, M.J., 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecol. 26, 32-46.
Associação dos Municípios do Oeste do Paraná – AMOP, 2018. Mapa Região da AMOP [online]. Retrieved in 2020, Jan 20, from http://www.amop.org.br/wp-content/uploads/2018/05/MAPA.pdf

Barnard, S., Morgenthal, T.L., Stolz, M., & Venter, A., 2021. Impact of land-use and flow conditions on the phytoplankton of the Sabie River, South Africa. Bothalia 51(1), a6. http://dx.doi.org/10.38201/btha.abc.v51.i1.6.

Bicudo, C.E.M. & Menezes, M., 2017. Gêneros de algas de águas continentais do Brasil: chave para identificação e descrições. São Carlos: RiMa.

Bohnenberger, J.E., Schneck, F., Crosetti, L.O., Lima, M.S., & Motta-Marques, D.D., 2018. Taxonomic and functional nestedness patterns of phytoplankton communities among coastal shallow lakes in southern Brazil. J. Plankton Res. 40(5), 555-567. http://dx.doi.org/10.1093/plankt/fby032

Bolgovics, A., Várbiró, G., Ács, E., Trábert, Z., Kiss, K.T., Pozdkerka, É.V., Görgényi, J., Boda, P., Lukács, B.A., Nagy-László, Z., Abonyi, A., & Borics, G., 2017. Phytoplankton of rithral rivers: its origin, diversity and possible use for quality-assessment. Ecol. Indic. 81, 587-596. http://dx.doi.org/10.1016/j.ecolind.2017.04.052.

Borcard, D., Gillet, F., & Legendre, P., 2011. Numerical Ecology with R. New York: Springer. http://dx.doi.org/10.1007/978-1-4419-7976-6.

Boric, G., Abonyi, A., Salmaso, N., & Ptacnik, R., 2021. Freshwater phytoplankton diversity: models, drivers and implications for ecosystem properties. Hydrobiologia 848(1), 53-75. PMid:32836348. http://dx.doi.org/10.1007/s10750-020-04332-9

Bortolini, J.C., Rodrigues, L.C., Jati, S., & Train, S., 2014. Phytoplankton functional and morphological groups as indicators of environmental variability in a lateral channel of the Upper Paraná River floodplain. Acta Limnol. Bras. 26(1), 98-108. http://dx.doi.org/10.1590/S2179-975X2014000100011.

Brasil, J., & Huszar, V.L.M., 2011. O papel dos traço funcionais na ecologia do fitoplâncton continental. Oecol. Aust. 15(4), 799-834. http://dx.doi.org/10.4257/ecoce.2011.1504.04.

Bussi, G., Whitehead, P.G., Bowes, M.J., Read, D.S., Prudhomme, C., & Dadson, S.J., 2016. Impacts of climate change, land-use change and phosphorus reduction on phytoplankton in the River Thames (UK). Sci. Total Environ. 572, 1507-1519. PMid:26927961. http://dx.doi.org/10.1016/j.scitotenv.2016.02.109.

Carlson, R.E., 1977. A trophic state index for lakes. Limnol. Oceanogr. 22(2), 361-369. http://dx.doi.org/10.4319/lo.1977.22.2.0361.

Casé, M., Leça, E.E., Leitão, S.N., Sant’Anna, E.E., Schwamborn, R., & de Moraes Junior, A.T., 2008. Phytoplankton community as an indicator of water quality in tropical shrimp culture ponds. Mar. Pollut. Bull. 56(7), 1343-1352. PMid:18538353. http://dx.doi.org/10.1016/j.marpolbul.2008.02.008.

Castro-Roa, D., & Pinilla-Agudelo, G., 2014. Periphytic diatom index for assessing the ecological quality of the Colombian Andean urban wetlands of Bogotá. Limnmetica 33, 297-312.

Conceição, L.P., Afife, H.J.M., Silva, D.M.L., & Nunes, J.C.M., 2021. Spatio-temporal variation of the phytoplankton community in a tropical estuarine gradient, under the influence of river damming. Reg. Stud. Mar. Sci. 43, 101642. http://dx.doi.org/10.1016/j.rsma.2021.101642.

Cordoba, J., Perez, E., Van Vlierberghge, M., Bertrand, A.R., Lupo, V., Cardol, P., & Baurain, D., 2021. De novo transcriptome meta-assembly of the mixotrophic freshwater microalga *Euglena gracilis*. Genes (Basel) 12(6), 842. PMid:34072576. http://dx.doi.org/10.3390/genes12060842.

Cupertino, A., Gucker, B., Von Rücker, G., & Figueredo, C.C., 2019. Phytoplankton assemblage composition as an environmental indicator in routine lentic monitoring: taxonomic versus functional groups. Ecol. Indic. 101, 522-532. http://dx.doi.org/10.1016/j.ecolind.2019.01.054.

Descy, J.P., Leitao, M., Everbecq, E., Smits, J.S., & Deliège, J.F., 2011. Phytoplankton of the River Loire, France: a biodiversity on modelling study. J. Plankton Res. 34(2), 120-135. http://dx.doi.org/10.1093/plankt/fbr085.

Domingues, C.D., & Torgan, L.C., 2012. Chlorophyta de um lago artificial hipereutrófico no sul do Brasil. Ilheringia 67, 75-91.

Fietz, S., Kobanova, G., Izmost’eva, L., & Nicklisch, A., 2005. Regional, vertical and seasonal distribution of phytoplankton and photosynthetic pigments in lake Baikal. J. Plankton Res. 27(8), 793-810. http://dx.doi.org/10.1093/plankt/fbi054.

Gleick, P.H., 2018. Transitions to freshwater sustainability. Perspective PNAS, 115(36), 8863-8871. https://doi.org/10.1073/pnas.1808893115.

Graco-Roza, C., Soininen, J., Corrêa, G., Pacheco, F.S., Miranda, M., Domingos, P., & Marinho, M.M., 2021. Functional rather than taxonomic diversity reveals changes in the phytoplankton community of a large dammed river. Ecol. Indic. 121, 107048. http://dx.doi.org/10.1016/j.ecolind.2020.107048.

Instituto Brasileiro de Geografia e Estatística – IBGE, 2010. Censo demográfico [online]. Retrieved in 2021, Jul 18, from https://www.ibge.gov.br/pt/incipio.html

Jia, J., Gao, Y., Song, X., & Chen, S., 2019. Characteristics of phytoplankton community and water net primary productivity response to the nutrient status of the poyang lake na gan river, China. Ecohydrology 12(7), 2136. http://dx.doi.org/10.1002/eco.2136.

Silva, T.T. et al.
Comunidade fitoplanctônica como discriminador ambiental em um trecho do rio salgado, semiárido nordestino. Cad. Cult. Cienc. 15(2), 29-41. http://dx.doi.org/10.14295/cad.cult.cienc.v15i2.1146.

Rao, C.R., 1964. The use and interpretation of principal componente analysis in Applied research. Sankhya 26, 329-358.

Reynolds, C.S., 2006. Ecology of phytoplankton. Cambridge: Cambridge University Press.

Reynolds, C.S., Elliot, J.A., & Frassl, M.A., 2014. Predictive utility of trait-separated phytoplankton groups: a robust approach to modeling population dynamics. J. Great Lakes Res. 40(3), 143-150. http://dx.doi.org/10.1016/j.jglr.2014.02.005.

Rodrigues, S.C., Torgan, L., & Schwarzbold, A., 2007. Composição e variação sazonal da riqueza do fitoplâncton no foz de rios do delta do Jacuí, RS. Brasil. Acta Bot. Bras. 21(3), 707-721. http://dx.doi.org/10.1590/S0102-33062007000300017.

Round, F.E., 1965. The biology of the algae. London: Edward Arnold.

Round, F.E., 1971. The taxonomy of the Chlorophyta, 2. Brit. J. Phycol. 6, 235-26.

Salmaso, N., & Tolotti, M., 2021. Phytoplankton and anthropogenic changes in pelagic environments. Hydrobiologia 848(1), 251-284. http://dx.doi.org/10.1007/s10750-020-04323-w.

Salmaso, N., Naselli-Flores, L., & Padišák, J., 2015. Functional classifications and their application in phytoplankton ecology. Freshw. Biol. 60(4), 603-619. http://dx.doi.org/10.1111/fwb.12520.

Santana, L.M., Moraes, M.E.B., Silva, D.M.L., & Ferragut, C., 2016. Spatial and temporal variation of phytoplankton in a tropical eutrophic river. Braz. J. Biol. 76(3), 600-610. PMid:27097084. http://dx.doi.org/10.1590/1519-6984.18914.

Santos, L.C.R., Lima, S.A., Cavalcanti, B.E., Melo, M.C., & Marques, N.M., 2018. Aplicação de índices para avaliação da qualidade da água da bacia costeira da sapucaia em Sergipe. Eng. Sanit. Ambient. 23(1), 33-46. http://dx.doi.org/10.1590/s1413-41522017159832.

Schulz, U.H., & Martins-Junior, H., 2001. Astyanax fasciatus as bioindicator of water pollution of Rio dos Sinos, RS. Braz. J. Biol. 61(4), 615-622. PMid:12071317. http://dx.doi.org/10.1590/S1519-69842001000400010.

Shi, Y., Eissenstat, D.M., He, Y., & Davis, K.J., 2017. Using a spatially-distributed hydrologic biogeochemistry model with nitrogen transport to study the spatial variation of carbon stocks and fluxes in a Critical Zone Observatory. In: American Geophysical Union Fall Meeting. New Orleans, LA.: Critical Zone Observatories, Dec. 11-15.

Silva, S.C.A., Farias, N.S.N., & Pereira-Junior, A., 2020. Diatomáceas como indicadoras da qualidade da água em rios urbanos. Braz. J. Dev. 6(6), 34616-34643. http://dx.doi.org/10.34117/bjdv6n6-125.

Simić, S.B., Karadžić, V.R., Cavijan, M.V., Vasiljević, B.M., Miličić, R., Šćančar, J., & Paunović, M., 2015. Comunidades de Algal ao longo do rio Sava, The Sava River. Berlin Heidelberg: Springer. 229-248.

Soofiani, N.M., Hatami, R., Hemami, M.R., & Ebrahimi, E., 2012. Effects of trout farm effluent on water quality and the macrobenthic invertebrate community of the Zayande-Roud River, Iran. N. Am. J. Aquaculture 74(2), 132-141. http://dx.doi.or g/10.1080/15222055.2012.672367.

Species Link, 2021 [online]. Retrieved in 2021, Jul 18, from www.splink.cria.org.br

Sun, J., & Liu, D., 2003. Geometric models for calculating cell biovolume and surface area for phytoplankton. J. Plankton Res. 25(11), 1331-1346. http://dx.doi.org/10.1093/plankt/fbg096.

Toledo, A.P., Talarico, M., Chinez, S.J., & Agudo, D., 1983. Aplicação de modelos simplificados para a avaliação de processos de eutrofização em lagos e reservatórios tropicais. In: Anais do Congresso Brasileiro de Engenharia Sanitária, Rio de Janeiro: ABES, 1-34.

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. http://dx.doi.org/10.1007/s10750-012-1201-2.

Trindade, R.M.L., Santos, S.M., Souza, C.A., Santos, C.R.A., & Bortolini, J.C., 2021. Using morphofunctional characteristics as a model of phytoplankton dynamics in a tropical reservoir. Braz. J. Bot. 44(2), 467-477. http://dx.doi.org/10.1007/s40415-021-00705-z.

Uhelinger, V., 1964. Étude statistique des methods de dénobrement planctonique. Arch. Sci. 17, 121-223.

Utermöhl, H., 1958. Zur Vervollkommung der quantitativen Phytoplankton-Methodik. Int. Vereinigung Theoretische Angew. Limnol. Mitt. 9, 1-38.

Vörösmarty, C.J., McIntyre, P.B., Gessner, M.O., Dudgeon, D., Prusevich, A., Green, P., Gliddon, S., Bunn, S.E., Sullivan, C.A., Liermann, C.R., & Davies, P.M., 2010. Global threats to human water security and river biodiversity. Nature 467(7315), 555-561. PMid:20882010. http:// dx.doi.org/10.1038/nature09440.

Wan, R., Cai, S., Li, H., Yang, G., Li, Z., & Nie, X., 2014. Inferring land use and land cover impact on stream water quality using a Bayesian hierarchical modeling approach in the Xitiaoxi River Watershed, China. J. Environ. Manage. 133, 1-11. PMid:24342905. http://dx.doi.org/10.1016/j.jenvman.2013.11.035.

Wang, C., Jia, H., Wei, J., Yang, W., Gao, Y., Liu, Q., Ge, D., & Wu, N., 2021. Phytoplankton
functional groups as ecological indicators in a subtropical estuarine river delta system. Ecol. Indic. 126, 107-165. http://dx.doi.org/10.1016/j.ecolind.2021.107651.

Xiao, J., Wang, L., Deng, L., & Jin, Z., 2019. Characteristics, sources, water quality and health risk assessment of trace elements in river water and well water in the Chinese Loess Plateau. Sci. Total Environ. 650(Pt 2), 2004-2012. PMid:30290343. http://dx.doi.org/10.1016/j.scitotenv.2018.09.322.

Yang, J., Wang, F., Lv, J., Liu, Q., Nan, F., Liu, X., Xu, L., Xie, S., & Feng, J., 2019. Interactive effects of temperature and nutrients on the phytoplankton community in an urban river in China. Environ. Monit. Assess. 191(11), 688. PMid:31664528. http://dx.doi.org/10.1007/s10661-019-7847-8.

Yang, J.R., Yu, X.Q., Chen, H.H., Kuo, Y.M.M., & Yang, J., 2021. Structural and functional variations of phytoplankton communities in the face of multiple disturbances. J. Environ. Sci. (China) 100, 287-297. PMid:33279042. http://dx.doi.org/10.1016/j.jes.2020.07.026.

Yang, M., Xia, J., Cai, W., Zhou, Z., Yang, L., Zhu, X., & Li, C., 2020. Seasonal and spatial distributions of morpho-functional phytoplankton groups and the role of environmental factors in a subtropical river-type reservoir. Water Sci. Technol. 82(11), 2316-2330. PMid:33339787. http://dx.doi.org/10.2166/wst.2020.489.

Yu, D., Shi, P., Liu, Y., & Xun, B., 2013. Detecting land use-water quality relationships from the viewpoint of ecological restoration in an urban area. Ecol. Eng. 53, 205-216. http://dx.doi.org/10.1016/j.ecoleng.2012.12.045.

Zhang, Y., Peng, C., Huang, S., Wang, J., Xiong, X., & Li, D., 2019. The relative role of spatial and environmental processes on seasonal variations of phytoplankton beta diversity along different anthropogenic disturbances of subtropical rivers in China. Environ. Sci. Pollut. Res. Int. 26(2), 1422-1434. PMid:30426374. http://dx.doi.org/10.1007/s11356-018-3632-4.

Zhang, Z., Gao, J., & Cai, Y., 2020. The direct and indirect effects of land use and water quality on phytoplankton communities in an agriculture-dominated basin. Environ. Monit. Assess. 192(12), 760. PMid:33184779. http://dx.doi.org/10.1007/s10661-020-08728-x.

Zhao, C., Liu, C., Xia, J., Zhang, Y., Yu, Q., & Eamus, D., 2012. Recognition of key regions for restoration of phytoplankton communities in the Huai River basin, China. J. Hydrol. 420-421, 292-300. https://doi.org/10.1016/j.jhydrol.2011.12.016.

Zohary, T., Flaim, G., & Sommer, U., 2021. Temperature and the size of freshwater phytoplankton. Hydrobiologia 848(1), 143-155. http://dx.doi.org/10.1007/s10750-020-04246-6.

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