JOURNAL OF WATER AND LAND DEVELOPMENT

e-ISSN 2083-4535

Polish Academy of Sciences (PAN) Institute of Technology and Life Sciences – National Research Institute (ITP – PIB)

JOURNAL OF WATER AND LAND DEVELOPMENT
DOI: 10.24425/jwld.2024.151807
2024, No. 63 (X-XII): 208–221

Causes of eutrophication in small water reservoirs in urban areas

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RECEIVED 31.07.2024 ACCEPTED 28.10.2024 AVAILABLE ONLINE 23.12.2024

Highlights

- The causes of eutrophication of urban reservoirs were investigated.
- River water rich in biogenic substances fertilise the ponds more than groundwater.
- Diatoms but not cyanobacteria dominate in the phytoplankton in hypertrophic ponds.
- It is necessary to reduce nutrients from the river through biogeochemical barriers.

Abstract: The aim of the study was to determine the causes of eutrophication in small urban water reservoirs located in the UMCS Botanical Garden in Lublin, supplied via surface and groundwater. The research (hydrological, hydrobiological, and hydrochemical), which included both field and laboratory work, was conducted during the growing season in the years 2022–2023. These ponds are fed by waters from the Czechówka River and, to a lesser extent, by groundwater (seepage). Both river and groundwater are characterised by high concentrations of mineral forms of nitrogen and phosphorus, ranging from 1.49 to 12.0 mg N·dm^{-3} and 0.07 to 0.21 mg P·dm^{-3}, respectively. This contributes to the intensive development of phytoplankton, especially during the summer period, with diatoms dominating the phytoplankton structure. The trophic state of the ponds ranges from eutrophic to hypertrophic. The study showed that the high degree of eutrophication was due to the load of nutrients delivered by the waters of the Czechówka River. Despite having several times higher concentrations of mineral and total nitrogen than in the river water, the spring water feeding the ponds had a negligible impact on the quality of the pond water due to low flow rate (<0.5 dm^3·s^{-1}). Therefore, the construction of urban ponds as part of green-blue infrastructure should consider the possibility of reducing nutrients through biogeochemical barriers and suspended sedimentation. It is also advisable to partially shade the water surface by planting trees in the shoreline zone to limit water heating and phytoplankton development.

Keywords: city ponds, diatoms, eutrophication, phytoplankton, water quality

INTRODUCTION

Urban ponds constitute significant freshwater resources (Oertli *et al.*, 2009). They also provide several landscape functions within urban areas, contribute to the increase of biodiversity, regulate the local microclimate and act as reservoirs for rainwater (Robitu *et al.*, 2006; Gledhill, James and Davies, 2008; Downing, 2010; Hassall, 2014; Oertli and Parris, 2019). Additionally, they provide ecosystem services for city residents, including social, recreational, cultural and educational benefits (Bolund and Hunhammar, 1999; Gledhill and James, 2012; Ghermandi and Fichtman, 2015; Hill *et al.*, 2017). In urban space management, they are elements of the blue-green infrastructure (BGI) (Krivtsov *et al.*, 2022). Although urban ponds play significant roles for both city residents and biodiversity, they are highly susceptible to anthropogenic impacts leading to water quality degradation (Tixier *et al.*, 2011). Their relatively high susceptibility to eutrophication (Roijackers, Aalderink and Blom, 1998; Smith and Schindler, 2009) is due to their morphometric characteristics. Most of these urban water bodies are small, shallow and have stagnant water (Brönmark and Hansson, 2002).

The main cause of eutrophication in urban ponds is the excessive influx of nutrients (Waajen, Faassen and Lürling, 2014). Nutrients entering these types of water bodies originate from various sources, including treated and untreated domestic sewage, street pollution and bird droppings (Scherer *et al.*, 1995; Stoianov, Chapra and Maksimovic, 2000; Waschbusch, Selbig and Bannerman, 2000). Contaminants are also carried to ponds by wind, rainwater or surface runoff (Müller *et al.*, 2020). Additionally, fertilisers used on lawns and organic matter from feeding fish and birds can enter these waters (Smith and Schindler, 2009; Chaichana, Leah and Moss, 2011). Besides the external nutrient load, bottom sediment, which act as an internal source of nutrients, can play a significant role in eutrophication (Søndergaard, Jensen, and Jeppesen, 2003; Zamparas and Zacharias, 2014). Moreover, large populations of fish inhabiting ponds can exacerbate the problem of eutrophication (Peretyatko *et al.*, 2009).

As a consequence of the excessive nutrient load entering ponds, intense eutrophication occurs, accompanied by harmful phytoplankton blooms. These blooms can cause hypoxia, fish kills and significant water turbidity (Pearl *et al.*, 2001; Scheffer, 2004; Waajen, Faassen and Lürling, 2014; Casa *et al.*, 2020). The excessive oxygen consumption by heterotrophic bacteria decomposing the blooming phytoplankton leads to oxygen depletion at the bottom of the ponds, creating dead zones for fish and other oxygen-dependent animals (Tittmann, 2024). Moreover, the high density of phytoplankton reduces water transparency. This results in vertical light attenuation and the inhibition of the growth of periphyton and submerged macrophytes (Han and Cui, 2016). Eutrophication often leads to cyanobacterial blooms that produce toxic substances, posing a threat to wildlife and water users (Peretyatko *et al.*, 2010; Waajen *et al.*, 2016; Tilahun *et al.*, 2019). Consequently, many global studies on urban ponds focus on assessing microcystin levels (Lürling and Faassen, 2012; Waajen, Faassen and Lürling, 2014; Prasertphon, Jitichum and Chaichana, 2020). In cases of severe eutrophication and weather anomalies, blooms of other algal groups, such as diatoms, are also observed (Casa *et al.*, 2020).

In addition to high nutrient content, urban ponds are often surrounded by buildings, trees, and shrubs that give wind protection. Low wind speeds promote stratification, which in turn favours algal blooms (Condie and Webster, 2001). Over the years, numerous attempts have been made to control or mitigate eutrophication in urban ponds through various geo-engineering measures such as surface aerator installation, sediment removal and biomanipulation (Waajen *et al.*, 2016; Hao *et al.*, 2021).

In Poland, urban ponds are relatively under-researched water bodies. For these small ponds functioning within city spaces, there are only a few studies available that document phycological aspects or trophic states (Messyasz and Jurgońska, 2003; Wołowski and Kowalska, 2009; Jekatierynczuk-Rudczyk, Zieliński and Puczko, 2016; Richter and Bączek, 2016). Much more attention is given to cyanobacteria and algae in fish ponds due to their importance for fish farming (Burchardt, Messyasz and Mądrecka, 2006; Bucka and Wilk-Woźniak, 2007; Napiórkowska-Krzebietke, Hutorowicz and Tucholski, 2011). Another relatively common topic is the implementation of reclamation activities in urban ponds (Wiśniewski, 2007; Kozak *et al.*, 2017; Jurczak *et al.*, 2018; Widelska and Walczak, 2020).

The aim of this study was to identify the causes of eutrophication in small urban ponds, using the ponds located in the UMCS Botanical Garden in Lublin as a case study. Understanding the processes responsible for their eutrophication will enable the development of methods to mitigate adverse trophic changes in such water bodies in the subsequent stage of research.

MATERIALS AND METHODS

STUDY AREA

The research subjects were two small water reservoirs located within the city of Lublin (Eastern Poland) – Figure 1. These reservoirs are situated within the administrative boundaries of the Maria Curie-Skłodowska University Botanical Garden (UMCS Botanical Garden), which covers an area of 21.25 ha. Physiographically, the ponds are located in the central valley of the Czechówka River, at the base of a loess slope intersected by three

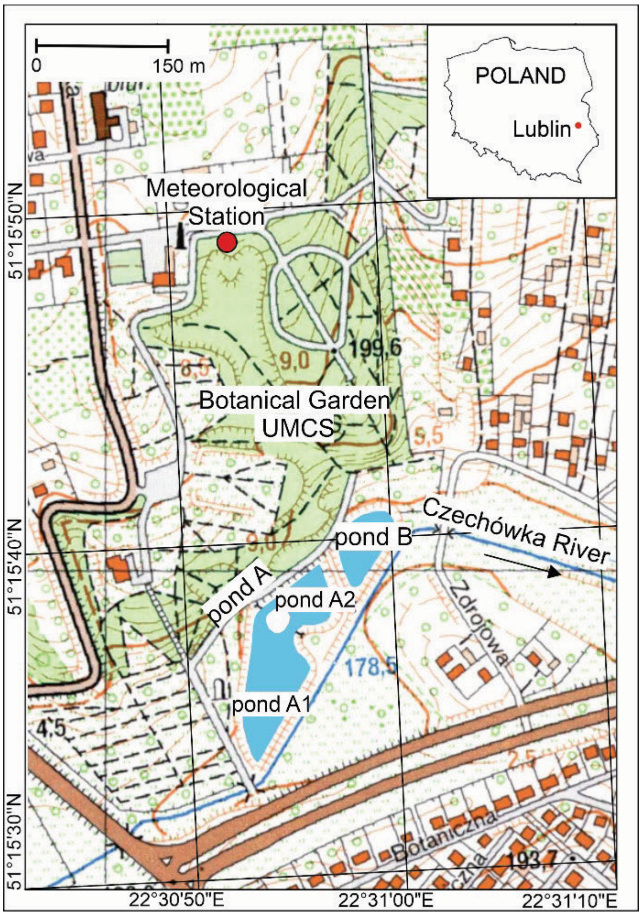


Fig. 1. Location of the ponds in the Botanical Garden of Maria Curie-Skłodowska University in Lublin; source: own study based on GUGiK (2001)

gullies (north-western shorelines of the ponds). The sloping terrace of the slope is covered with species such as *Acer platanoides* L., *Aesculus hippocastanum* L., *Carpinus betulus* L., *Fraxinus excelsior* L., *Robinia pseudoacacia* L., *Sambucus nigra* L., *Tilia cordata* Mill., and *Ulmus minor* Mill. (Dąbrowska, 2014). To the south, the ponds border an expressway (Aleja Solidarności), to the southwest, a city single-lane street (Aleja Warszawska), and from the remaining sides, they are bordered by residential neighbourhoods of single-family homes.

The analysed reservoirs obtained their current shape at the turn of the 20th and 21st centuries. Revitalisation efforts were conducted in 1996 and 2000, during which the ponds were lined with a polypropylene mat filled with sodium bentonite (Kwiatkowski, 1997). The reservoirs form a complex of two ponds (A and B), hydraulically connected to each other by an underwater siphon. Their total area is slightly over 1 ha (Tab. 1). The pond located in the southern part of the Botanical Garden (pond A) is more than twice as large and features an artificial island. Morphometrically, pond A consists of two basins connected by two narrow channels. Therefore, analyses and observations were conducted separately for both parts (pond A1 and pond A2). Pond A also has three times the water volume of pond B, although their average depths are almost identical (about 0.7 m). The maximum depth was recorded in pond A at 1.45 m (Fig. 2). Additionally, the average thickness of sediments deposited in pond A was 0.20 m, while in pond B it was almost twice as much.

The ponds are primarily fed by the waters of the Czechówka River, a left-bank tributary of the Bystrzyca River, which flows into the Wieprz River (a right-bank tributary of the Vistula River). Groundwater recharge and surface runoff have a significantly smaller contribution to the water balance of the ponds. The total length of the Czechówka River is 18 km, with an average flow rate of 160 $dm^3 \cdot s^{-1}$ (in the mouth of the river), and it shows a strong influence of anthropogenic pressure along its course (Michalczyk, Chmiel, and Głowacki, 2012). The waters of the Czechówka that feed the analysed ponds in the Botanical Garden are not typical river waters – their quality and quantity are modified by the operation of a flow-through pond complex located in its upper course (near The Lublin Open Air Village Museum, Pol.: Muzeum Wsi Lubelskiej). The gauging station profile located on the Czechówka at the level of the Botanical Garden of UMCS encompasses a catchment area of 61 km^2, which constitutes 75% of its river basin area (Michalczyk *et al.*, 2018).

Table 1. Basic morphometric parameters of the basins of ponds A and B in the Botanical Garden of Maria Curie-Skłodowska University and the sediments filling them (studies conducted on 26 Apr 2022 and 16 May 2022)

Parameter	Pond A (A1 and A2)	Pond B
Surface area (ha)	0.73	0.27
Water storage (m^3)	5880	1928
Mean depth (m)	0.72	0.73
Maximum depth (m)	1.45	1.26
Length of shoreline (m)	461	219
Mean sediment thickness (m)	0.20	0.38

Source: own study.

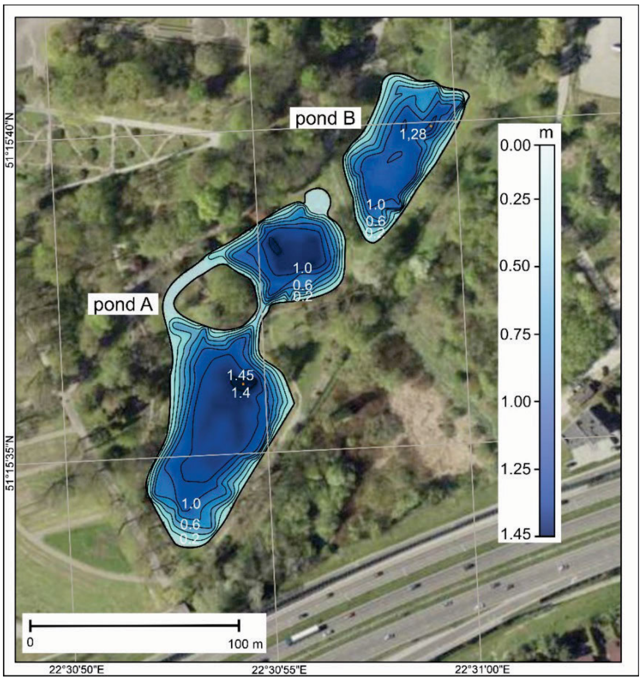


Fig. 2. Bathymetric plan of the ponds in the Botanical Garden of Maria Curie-Skłodowska University in Lublin; source: own study

Botanically, these are macrophytic reservoirs maintained in the form of wild park ponds, characterised by a sparse floral composition. The littoral zone is populated by species such as *Alisma plantago-aquatica* (L.), *Carex acutiformis* Ehrh, *Glyceria maxima* (Hartm.) Holmb, *Phragmites australis* (Cav.) Trin. ex Steud., *Rorippa amphibia* (L.) Besser, *Scirpus sylvaticus* L., *Sparganium erectum L. emend. Rchb. s. str., Typha angustifolia* L., *Iris pseudacorus* L., *Juncus effusus* L., *Rumex hydrolapathum* Huds. (Dąbrowska, 2014; Dąbrowska, Chernetskyy and Szymczak, 2015). Grasses dominate the shores, while the water surface is covered by free-floating plant species from the *Lemnetea* R. Tx. class, including *Lemna minor* L., *L. trisulca* L., *Spirodela polyrhiza* (L.) Schleid. as well as *Nuphar lutea* (L.) Sibth. & Sm., *Nymphaea alba* L. and *Trapa natans* L. s. l. In the summer season, the second reservoir (pond B) additionally exhibits very intense growth of free-floating species, which occupy almost the entire surface of its water (dominated by plants of the *Lemnetea* class and *Spirodela* genus). Submerged species in the analysed ponds include *Ceratophyllum demersum* L. s. str., *Elodea canadensis* Michx., *Myriophyllum spicatum* L. and *Potamogeton crispus* L. (Dąbrowska, 2014; Dąbrowska, Chernetskyy and Szymczak, 2015). The flora of the ponds includes several expansive plant species such as *Glyceria maxima*, *Lemna minor*, *Myriophyllum spicatum*, *Phragmites australis*, *Potamogeton crispus*, *Spirodela polyrhiza*, *Typha angustifolia* and *T. latifolia* L. Names of species are given according to Mirek *et al.* (2002).

The analysed reservoirs are not intended for fish farming purposes. Their high fertility leads to degradation of water quality.

FIELD RESEARCH

The water quality of the ponds in the UMCS Botanical Garden was studied during the years 2022–2023. Water samples in 2022 were collected from March to July, every 5–6 weeks, from four sampling points (Fig. 3): the Czechówka River feeding the pond

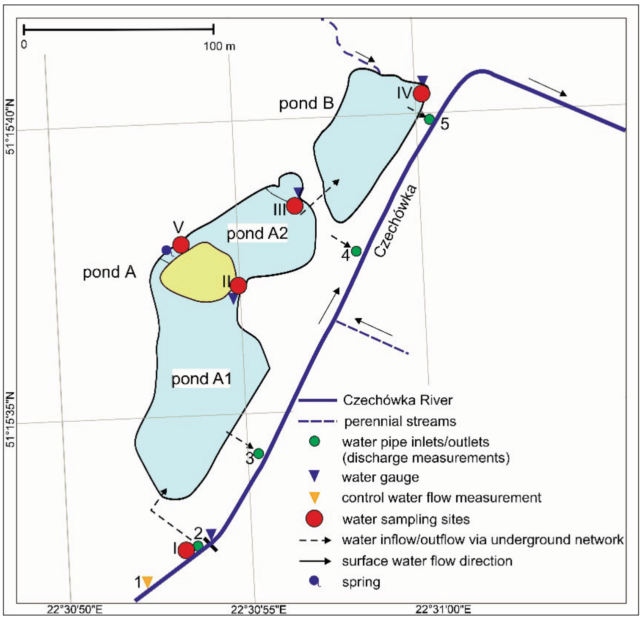


Fig. 3. Water circulation scheme in the ponds of the Botanical Garden of Maria Curie-Skłodowska University, including locations for water sampling and hydrological measurements; source: own study

(point no. I), pond A1 (point no. II), pond A2 (point no. III), and pond B (point no. IV) at the outflow to the Czechówka River. In 2023, water samples were taken from the same locations from March to October, at 5–6 week intervals. In 2023 (May, July, and September), water samples were also collected from the spring feeding pond A2 – point no. V (Fig. 3).

In all designated points on the ponds and river, the following measurements were conducted: water temperature, pH, electrical conductivity (EC), dissolved oxygen, turbidity (YSI 600XL Sonde). Total chlorophyll "a" and cyanobacterial chlorophyll "a" were also measured (Algae Torch Sonde). Water samples were collected for hydrochemical analysis, as well as quantitative and qualitative phytoplankton analysis. Surface water sampling followed PN-EN ISO 5667-1 and PN-EN ISO 5667-3 standards (Polski Komitet Normalizacyjny, 2008, Polski Komitet Normalizacyjny, 2018-08). For phytoplankton analysis, both unconcentrated samples (quantitative analysis) and preliminary concentrated samples at 50 dm^3 volume on a plankton net (20 μm) (qualitative analysis) were collected. Samples were fixed with Lugol's solution (Chempur, 10%). All water samples were transported to the laboratory in a thermal box at 4°C, protected from sunlight exposure.

Hydrometric measurements in 2022 and 2023 were conducted on the same days as the hydrochemical and hydrobiological studies. Flow rate measurements were carried out using a Hega 1 type hydrometric mill (Fig. 3). Due to the lateral inflow of water from the Czechówka River into the ponds, flow measurements were taken both at the inlet to the ponds (control profile no. 2 – water inflow from the Czechówka via a concrete channel) and at the outlet (control profile no. 5 – water outflow via a concrete channel to the Czechówka) – Figure 3. Hydrological monitoring also included assessing periodically functioning water outlets from the ponds (control profiles no. 3 and 4) and procuring a measurement profile on the Czechówka (no. 1), where flow measurements were also performed (Fig. 3). Water levels and flow rates were recorded at the Botanical Garden gauging profile on the Czechówka (at an artificial water impoundment on a concrete weir located beyond control profile no. 2) and at the water sampling points on the ponds. Additionally to surface water monitoring, the yield of a spring located near pond A2 was measured. The meteorological conditions during the measurement period were determined based on daily air temperature, precipitation and relative humidity data from the meteorological station of the Department of Hydrology and Climatology UMCS, located in the northern part of the Botanical Garden. Based on these indicators, the monthly values of potential evaporation were calculated using the Ivanov formula (Bryś *et al.*, 2022).

LABORATORY AND COMPUTATIONAL RESEARCH

In the Hydrochemical Laboratory of the Department of Hydrology and Climatology at UMCS, the contents of anions and cations were determined using a Metrohm ion chromatograph (model MIC-3). For the determination of anions (Cl^-, NO_2^-, NO_3^-, SO_4^{2-}), a Metrosep A SUPP5 250 column was used, and for cations (Na^+, NH_4^+, K^+, Ca^{2+}, Mg^{2+}), a Metrosep C2 150 column was employed. Additionally, the water samples were analysed for total nitrogen (TN) and total phosphorus (TP), and their dissolved mineral forms were calculated (Nmin, Pmin), along with biological oxygen demand (BOD_5) and chemical oxygen demand (COD_{Cr}). Detailed information regarding the scope of the laboratory tests and methodologies can be found in Table 2.

Table 2. Scope of conducted analyses of water quality in the ponds of the Botanical Garden of Maria Curie-Skłodowska University in Lublin

Scope	Method	Apparatus	Norm	Standard solutions, reference materials
I. Field measurements				
Water flow ($m^3 \cdot s^{-1}$)	flow meter	Hega flow meter 2		
Temperature (°C)	electrometric	Sonde YSI 600XL	–	
Reaction (pH)				HI 7004; HI 7007; HI 7009
Dissolved oxygen (mg $O_2 \cdot dm^{-3}$)				
Oxygen saturation (% O_2)				HI 7040-2
EC ($\mu S \cdot cm^{-1}$)			–	YSI 3167; Environment Canada MISSIPPI-14

cont. Tab. 2

Scope	Method	Apparatus	Norm	Standard solutions, reference materials
Transparency (m)	Secchi disc		–	–
Turbidity (*FTU*)	optical	Sonde AquaFluor	–	–
Chlorophyll „a”(μg·dm^{-3})			–	–
Cyanobacteria (μg·dm^{-3})			–	–
II. Laboratory measurments				
BOD_5 (mg O_2·dm^{-3})	spectrophotometric	Pastel UV	–	–
COD_{Cr} (mg O_2·dm^{-3})			–	–
Total suspended solids (mg·dm^{-3})			–	–
Ammonium nitrogen (mg NH_4^+·dm^{-3})	ion chromatography	Metrohm MIC-3	PN-EN ISO 14911:2002	Environment Canada MISSIPPI-14 and ION-915
Nitrate nitrogen (mg NO_3^-·dm^{-3})			PN-EN ISO 10304-1:2009	
Nitrite nitrogen (mg NO_2^-·dm^{-3})			PN-EN ISO 10304-1:2009	
Nmin (mg N·dm^{-3})	calculation			
Total nitrogen (mg N·dm^{-3})	spectrophotometric	Hach DR 900	HACH 10071	nitrogen total standard solution (Supelco)
Total phosphorus (mg P·dm^{-3})	spectrophotometric	Hach DR 900	HACH 8190	phosphorus total standard solution (Supelco)
Orthophosphates (mg PO_4^{3-}·dm^{-3})			HACH 8048	–
Pmin (mg P·dm^{-3})	calculation		–	–
Silica, iron content (μg·dm^{-3})	mass spectrometry	ICP-MS	–	EnviroMAT ES-L-2 CRM EnviroMAT ES-H-2 CRM

Explanations: *EC*= electrolytic conductivity at 25°C, BOD_5 = biological oxygen demand; COD_{Cr} = chemical oxygen demand, Nmin = mineral nitrogen (nitrite + nitrate + ammonium) (mg N·dm^{-3}), Pmin = orthophosphates (mg P·dm^{-3}). Source: own study.

The trophic state of the analysed waters was calculated using the trophic state index (*TSI*) (Carlson, 1977; Kratzer and Brezonik, 1981). The trophic state was determined as the average value from metrics (Eqs. 1–4): total phosphorus concentration – TP (mg·dm^{-3}) – Equation (1), chlorophyll “a” – Chl a (μg·dm^{-3}) – Equation (2), Secchi disk visibility – SD (m) – Equation (3) and total nitrogen – TN (mg·dm^{-3}) – Equation (4):

$$TSI(\mathrm{TP}) = 14\ln(TP) + 4.15 \tag{1}$$

$$TSI(\mathrm{Chl}) = 9.81\ln(\mathrm{CHLa}) + 30.6 \tag{2}$$

$$TSI(\mathrm{SD}) = 60 - 14.41\ln(\mathrm{SD}) \tag{3}$$

$$TSI(\mathrm{TN}) = 14.43\ln(\mathrm{TN}) + 54.45 \tag{4}$$

The abundance and biomass of phytoplankton were established using an inverted microscope (Olympus CKX53 microscope with EP50 camera), in accordance with the guidelines for conducting field and laboratory studies of lake phytoplankton (Hurtowicz and Pasztaleniec, 2020).

The results were statistically analysed with one-way analysis of variance (ANOVA) tests. The significance of differences between means was determined using Duncan’s test at a significance level of $\alpha = 0.05$. Measurement points I–IV were analysed. The relationships (Pearson correlation coefficient) between the studied indicators of water quality in ponds of the Botanic Garden were also assessed. Significance was set at $p \leq 0.05$, $p \leq 0.01$ and $p \leq 0.001$. All statistic analysis utilised Statistica 13.3.

RESULTS AND DISCUSSION

POND ALIMENTATION CONDITIONS

The average daily air temperature in 2022 was 9.2°C, and in 2023 it was 10.1°C. These values were significantly higher compared to the long-term measurements (1991–2020) at the Lublin-Radawiec station (8.2°C). Negative average monthly air temperatures were recorded only in December 2022 and 2023, while in other months they were positive (Fig. 4). In the long-term period, negative average monthly air temperatures usually occurred in December, January and February. The highest average monthly air temperatures were recorded from May to August. The variability in air temperature was typical for a temperate climate zone; however, it is important to highlight the significant increase in air temperatures during the winter months (from December to March). The average relative humidity values were

72.4% in 2022 and 73.5% in 2023. The total precipitation in 2022 was 698 mm, and in 2023, it was 804 mm. The long-term average (1991–2020) total precipitation at the Lublin-Radawiec station was 601 mm. Therefore, these were wet years compared to average years. The highest monthly precipitation totals (>80 mm) occurred in July, August and December 2022, whereas in 2023, they occurred in January, July, and October. Potential evaporation was around 790 mm in the years studied. In 2022, it was about 100 mm higher than the total precipitation, while in 2023, the total precipitation and potential evaporation were at a similar level. In the summer months (May–August), water losses in ponds due to evaporation amounted to around 460 mm (with sums precipitation of 284 mm). In the winter months (December–March), evaporation losses were around 110 mm (with sums precipitation of 210 mm).

The average flow of the Czechówka River at a measurement profile on the Czechówka (no. 1) in 2022–2023 was 63 $dm^3 \cdot s^{-1}$, which was lower than during the period of 2003–2017 (Michalczyk *et al.*, 2018), when it averaged 88 $dm^3 \cdot s^{-1}$. The seasonal flow pattern showed a relatively even distribution (Fig. 4), with slightly lower flow during the summer half-year. The highest flow (101 $dm^3 \cdot s^{-1}$) during the measurement period was recorded in September 2022, while the lowest flow was in December 2022 (36 $dm^3 \cdot s^{-1}$). The collected hydrometric data indicate an unfavourable situation in terms of river water resources and pond feeding. Water withdrawal to replenish losses in the ponds mainly occurred during the growing season, when the Czechówka River had relatively low flow and intense evaporation was taking place from the ponds. Measurements showed that no less than 1/3 of the current flow was withdrawn from the Czechówka River. With a water withdrawal rate of 30 $dm^3 \cdot s^{-1}$, complete water exchange in the pond occurs within 3 days. In contrast, at a withdrawal rate of 50 $dm^3 \cdot s^{-1}$, exchange occurs in less than 2 days. Therefore, the ponds in OB should be classified as reolimnic reservoirs.

PHYSICO-CHEMICAL WATER PROPERTIES AND WATER TROPHIC STATUS

The results of measurements of physico-chemical properties of water taken from the ponds of the UMCS Botanical Garden in the years 2022–2023 are presented in Table S1. No significant differences ($p > 0.05$) were found between the measurement points for the physical and chemical properties of the water, except for turbidity, which was significantly higher at point I compared to the other locations. The average turbidity of the examined waters ranged from 9.0 to 24.1 NTU. The highest turbidity in both research seasons was observed in the waters of the Czechówka River (Tab. S1). High values of water turbidity and the total suspended solids (TSS) in the ponds were also noted during periods of intense phytoplankton growth, which is confirmed by the determined Pearson coefficient ($p \leq 0.01$, Tab. S2). As a result of receiving river water, organic and inorganic suspensions can significantly disrupt the balance of urban pond ecosystems (Ghosh, Roy and Siddique, 2021).

The examined waters, regardless of the measurement point, exhibited significant temperature variability, ranging from 4.0 to 22.2°C in 2022 and from 9.1 to 23.3°C in 2023. This phenomenon is typical for a temperate climate zone (Gizińska and Sojka, 2023). During the study period, the lowest water temperatures were recorded in March and the highest in the summer months (July/August).

The pH of the examined waters varied seasonally from neutral to slightly alkaline. Lower water temperatures corresponded to lower pH levels, whereas as water temperature increased, the water pH also increased. Meanwhile, the level of dissolved mineral compounds expressed by *EC* generally remained within the range of moderately mineralised waters (<1000 $\mu S \cdot cm^{-1}$) in most cases. The highest *EC* values, regardless of the measurement point, were recorded in March. Water in urban ponds may exhibit elevated *EC* (up to 3,000 $\mu S \cdot cm^{-1}$),

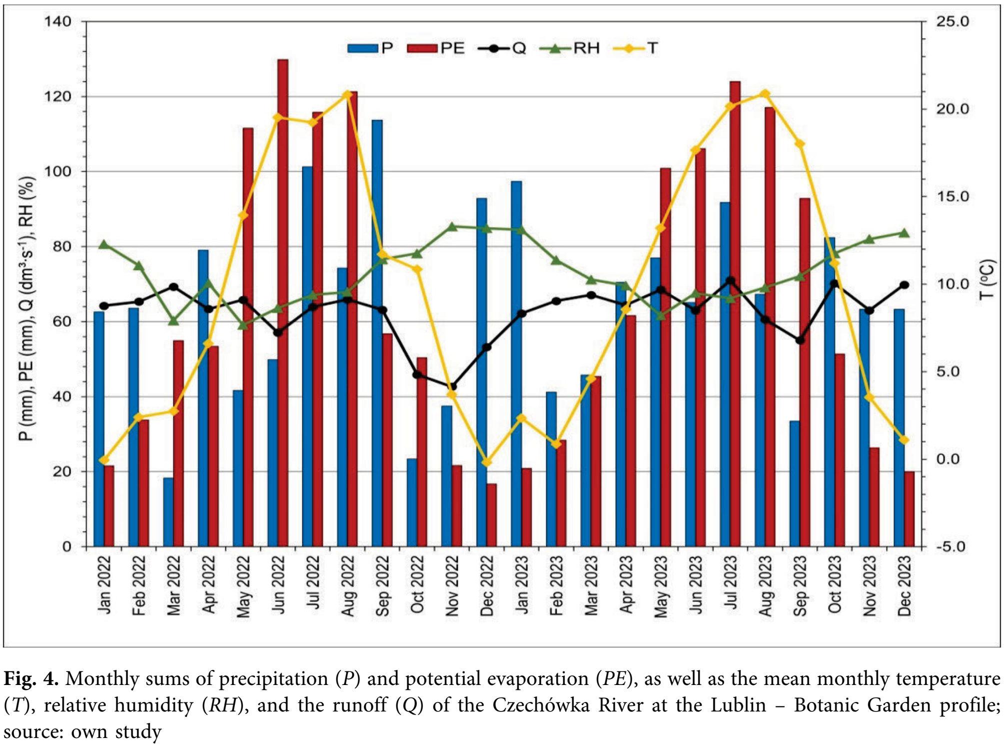


Fig. 4. Monthly sums of precipitation (*P*) and potential evaporation (*PE*), as well as the mean monthly temperature (*T*), relative humidity (*RH*), and the runoff (*Q*) of the Czechówka River at the Lublin – Botanic Garden profile; source: own study

resulting from the influx of pollutants such as road salt (Oertli and Parris, 2019). In such cases, *EC* can also serve as an indicator of water pollution levels. The significantly lower *EC* values obtained during the study suggest that the impact of anthropogenic factors on this parameter was less intense compared to other areas (Schagerl, Angeler and Biester, 2011).

The dissolved oxygen content of the studied waters showed a relatively large amplitude of seasonal changes, ranging from 1.13 to 21.6 mg $O_2 \cdot dm^{-3}$. In most cases, the lowest concentrations and oxygen saturation were recorded in the spring, while the highest were observed in the summer. Such changes are typical of eutrophic and hypertrophic reservoirs with an intense photosynthesis process (Nguyen, Bac and Hoang, 2016). An exception was the oxygen conditions in the waters of the Czechówka River and pond B in July 2022, which were extremely low (1.13–2.01 mg $O_2 \cdot dm^{-3}$, 12.9–24.5%). Additionally, in the summer of 2023, oxygen content was measured not only in the surface layer, but also in the bottom layer of the studied ponds. The results showed no significant changes in this parameter except for the oxygen conditions in pond A1. At this measurement point, the bottom layer had 22 to 35% lower oxygen content than the surface layer. During this period, the most intensive growth of phytoplankton occurred, which could have influenced the formation of oxygen stratification in the pond. In the surface layer, water is enriched with oxygen through the process of photosynthesis, while in the deeper layer, oxygen decreases due to its consumption in the decomposition of organic matter. At the bottom, no water movement was indicated at that time.

In the hydrochemical studies of the waters, special attention was given to the essential nutrients: nitrogen and phosphorus, and the related water trophic levels. The average content of mineral nitrogen and total nitrogen, regardless of the measurement point and research season, varied respectively from 1.65 to 2.18 mg $N \cdot dm^{-3}$ and from 3.06 to 4.4 mg $N \cdot dm^{-3}$. A similar range of total nitrogen content, from 1.4 to 3.3 mg $N \cdot dm^{-3}$, was recorded in urban ponds in the Netherlands (Waajen, Faassen and Lürling, 2014) and in Portugal from 1.5 to 4.46 mg $N \cdot dm^{-3}$ (Rodrigues *et al.*, 2022). A significantly higher level of total nitrogen (from 1.46 to 13.13 mg $N \cdot dm^{-3}$) was observed in urban ponds in Thailand (Prasertphon, Jitichum and Chaichana, 2020).

In the studied ponds, mineral nitrogen constituted an average of about 50–60% of the total nitrogen. In the pond waters, under favourable conditions for the development of phytoplankton, a significant decrease in its concentrations was observed, constituting only 6–20% of the total nitrogen, which was particularly evident in July and August 2023. Mineral nitrogen plays a very important role in the development of phytoplankton. The limitation of phytoplankton development due to nitrogen restriction is confirmed by Czech studies (Ivanova *et al.*, 2022). The content of mineral and total nitrogen in the waters of the Czechówka River was similar in the respective months to the nitrogen content in the ponds. Groundwater can provide a significant load of nitrogen to the ponds, but due to the low yield of the spring (<0.5 $dm^3 \cdot s^{-1}$) and the polypropylene lining of the bottoms and banks of the analysed ponds, they have a minimal impact on their eutrophication in this case. The concentration of mineral and total nitrogen in the spring feeding pond A2 was several times higher than in the waters of the Czechówka River and the pond. The proportion of mineral forms in the total nitrogen in the spring was about 85% (Tab. S1).

The average total phosphorus content, regardless of the measurement point and research season, ranged from 0.18 to 0.31 mg $P \cdot dm^{-3}$. This was comparable to results for urban water bodies in other parts of Europe: 0.16–0.44 mg $P \cdot dm^{-3}$ (Waajen, Faassen and Lürling, 2014), and significantly lower than in that within tropical countries, where it ranged from 0.10 to 34.8 mg $P \cdot dm^{-3}$ (Prasertphon, Jitichum and Chaichana, 2020). The proportion of mineral phosphorus in the total phosphorus pool was similar to that of nitrogen, at about 50%. An exception was the spring, where the proportion of mineral phosphorus was 95% of the total phosphorus. The content of mineral phosphorus was about 30% higher than in the waters of the Czechówka River. During the summer, the concentration of mineral phosphorus decreased in both the river and the pond waters. This phenomenon was particularly evident in July 2023, when a very high phytoplankton abundance was recorded.

The content of nitrogen and phosphorus in the studied waters and the proportion of their mineral forms indicate intense biological activity during the summer period, which consequently led to a decline in water quality (increased turbidity, BOD_5 and COD_{Cr}, and low transparency) – Table S1. The increase in the trophic potential of the pond water may also be influenced by the feeding of birds by visitors and the resuspension of bottom sediments, causing increased water turbidity (Ferreira *et al.*, 2022). The threshold concentrations of mineral nitrogen and phosphorus that cause intense phytoplankton development are estimated in some cases to be 0.3 mg $N \cdot dm^{-3}$ and 0.03 mg $P \cdot dm^{-3}$ (Chambers *et al.*, 2012). Based on the calculated limiting nutrient index – TN:TP (Waajen, Faassen and Lürling, 2014; Florida Lakewatch, 2000), it was determined that in the studied urban water bodies, in the 2022 season, nitrogen was the limiting nutrient (TN:TP ≤10), while in 2023, nitrogen and/or phosphorus (TN:TP 10–17) or phosphorus (TN:TP ≥17) were limiting. Nitrogen was also identified as the limiting nutrient in most urban ponds studied in Thailand (Prasertphon, Jitichum and Chaichana, 2020).

Chlorophyll "a" exhibited varying concentrations depending on the research season. In 2022, its content ranged from 32.6 to 63.0 $\mu g \cdot dm^{-3}$, while in 2023, it was significantly higher, ranging from 132.4 to 163.1 $\mu g \cdot dm^{-3}$. Despite this, regardless of the research season, the chlorophyll "a" concentrations were much lower than those observed in similar water bodies (Waajen, Faassen and Lürling, 2014; Prasertphon, Jitichum and Chaichana, 2020). No statistically significant differences were observed in the concentration of chlorophyll "a" in the ponds and in the waters of the Czechówka River (p > 0.05). The highest chlorophyll "a" concentrations were recorded in the summer months. Chlorophyll present in the waters of the studied ponds caused low water transparency, usually not exceeding 0.3 m, which was also observed in the studies by Waajen, Faassen and Lürling (2014).

For cyanobacterial chlorophyll, the average content in the 2022 season did not exceed 3.2 $\mu g \cdot dm^{-3}$, while in 2023 it was higher, ranging from 8.5 to 13.2 $\mu g \cdot dm^{-3}$. Cyanobacterial chlorophyll constituted only a few percent of the total chlorophyll, representing a minor share compared to the studies by Waajen, Faassen and Lürling (2014). The highest concentrations of cyanobacterial chlorophyll "a" were recorded in July 2022 and in July and September 2023.

The trophic state of the studied waters was assessed using the trophic state index (*TSI*) proposed by Carlson (1977) and supplemented by Kratzer and Brezonik (1981). A *TSI* level indicating oligotrophy is below 40, mesotrophy ranges from 40–50, eutrophy from 50–70, and hypertrophy above 70. The studied waters exhibited a high degree of eutrophication within the hypertrophic range (Fig. 5). The exception was the state in March 2022, where the *TSI* values fell within the eutrophic range, although the *TSI*(TN) and *TSI*(TP) values were at the hypertrophic level. In other months, all sampled points in the waters of the Czechówka River and the ponds of the Botanical Garden showed hypertrophy, indicating poor water quality. Very high *TSI* values were also obtained in studies of urban ponds in Argentina during strong diatom blooms (Casa *et al.*, 2020).

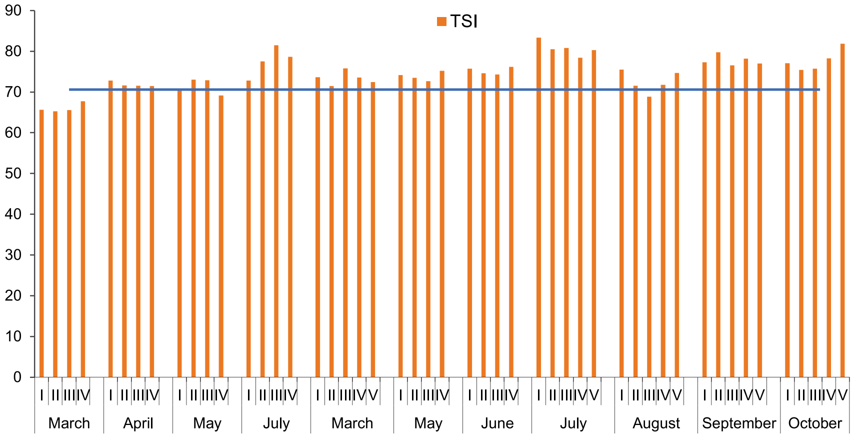


Fig. 5. Trophic status index (*TSI*) of the analysed waters in five study points – the Czechówka River (points I and V) and the ponds of the Botanical Garden of Maria Curie-Skłodowska University (points II, III and IV); source: own study

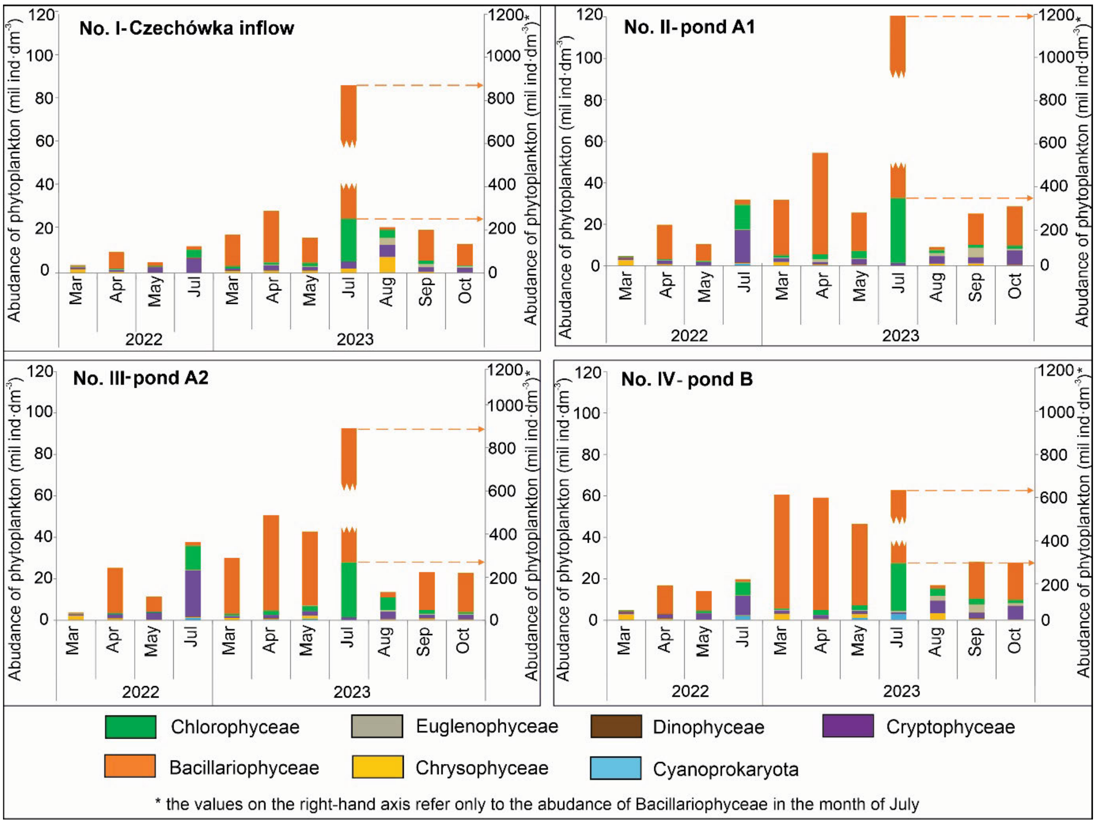
PHYTOPLANKTON

The species composition of the phytoplankton and its seasonal variability in both the pond waters and the river feeding them were similar. This situation may result from the fact that the Czechówka River, in its upper course, flows through a complex of ponds that alter the species composition of its phytoplankton. The small volume of the Botanical Garden ponds and the relatively fast rate of water exchange cause these ponds to be colonised by species transported by the river (Wetzel, 2001). Consequently, there are no significant differences in abundance, biomass and algal species composition between these ecosystems. This situation was also confirmed by ANOVA test ($p > 0.05$). Typically, the dominant group in the phytoplankton was *Bacillariophyceae* (Figs. 6 and 7). In spring 2023 (March, May), the maximum biomass of *Bacillariophyceae* exceeded 90 mg·dm^{-3} in pond B (Fig. 7), while the highest abundance of this group was in July 2023, reaching 876·10^{6} ind.·dm^{-3} (Fig. 6). The observed dominance of diatoms for most of the study period can be associated with the existing water movement in the ponds, resulting from the feeding by the Czechówka River. Diatoms prefer non-static, turbulent waters or periods of water circulation, as this facilitates their maintenance in the water column. During stable water periods, their heavy, silica shells contribute to their sedimentation at the bottom of the reservoirs (Kawecka and Eloranta, 1994; Wilk-Woźniak and Ligęza, 2003; Wang, Yang and Kattel, 2018). The dominance of diatoms was also found in the waters of urban ponds in Brussels that were fed by small rivulets and ground water seepage (Peretyatko *et al.*, 2009), as well as in other locations in Poland – ponds in Żywiec fed by the Soła River (Jachniak and Młyniuk, 2019).

Among *Bacillariophyceae*, the dominant species in both abundance and biomass was *Cyclotella cf. meneghiniana*. Only in July 2023, there was a mass occurrence of *Cyclotella cf. atomus*. These small centric diatoms significantly influenced the abundance in that month, although not translating into biomass. Species from the *Cyclotella* genus are characteristic of plankton in lowland rivers (*C. atomus*, *C. meneghiniana*) and eutrophic ponds (*C. meneghiniana*), characterised by high electrical conductivity (Witkowski *et al.*, 2010; Bąk *et al.*, 2012). Research by Liang *et al.* (2020) also indicates their high abundance in eutrophic small urban lakes in China.

The statistical analysis indicated a strong positive correlation ($p \leq 0.001$, Tab. S2) between phytoplankton abundance and chlorophyll "a" content. In our study, we observed maximum chlorophyll "a" values exceeding 330 μg·dm^{-3} (in the river), which coincided with periods of very high abundance of small centric diatoms (*C. atomus*). Similar relationships indicating a connection between blooms of centric diatoms and high chlorophyll "a" concentrations have been noted in studies conducted in Russia and China (Kolmakov *et al.*, 2002; Liang *et al.*, 2020). Japanese studies also confirm the dominant development of small-cell, centric species like *Cyclotella* with very high chlorophyll "a" concentrations (Amano, Takahashi and Machida, 2012). This phenomenon is linked to the efficient uptake and storage of nutrients by small cells due to their high surface-to-volume ratio. This ability was particularly evident in July 2023, characterised by reduced concentrations of mineral phosphorus during the period of intensive development of the species *Cyclotella cf. atomus*.

The development of diatoms in water depends on various environmental factors, including the availability of microelements, especially silicon, which is a key nutrient limiting their growth (Florida Lakewatch, 2000; Sumper and Kröger, 2004; Brzezinski *et al.*, 2011). In our study in 2022 (Tab. S1), we observed only a slight decrease in silicon content in the pond waters, suggesting that the summer (July) limitation of diatom growth was not



* the values on the right-hand axis refer only to the abudance of Bacillariophyceae in the month of July

Fig. 6. Abundance of phytoplankton (mln ind.·dm^{-3}) in four study points – the Czechówka River (point I) and the ponds of the Botanical Garden of Maria Curie-Skłodowska University (points II, III and IV); source: own study

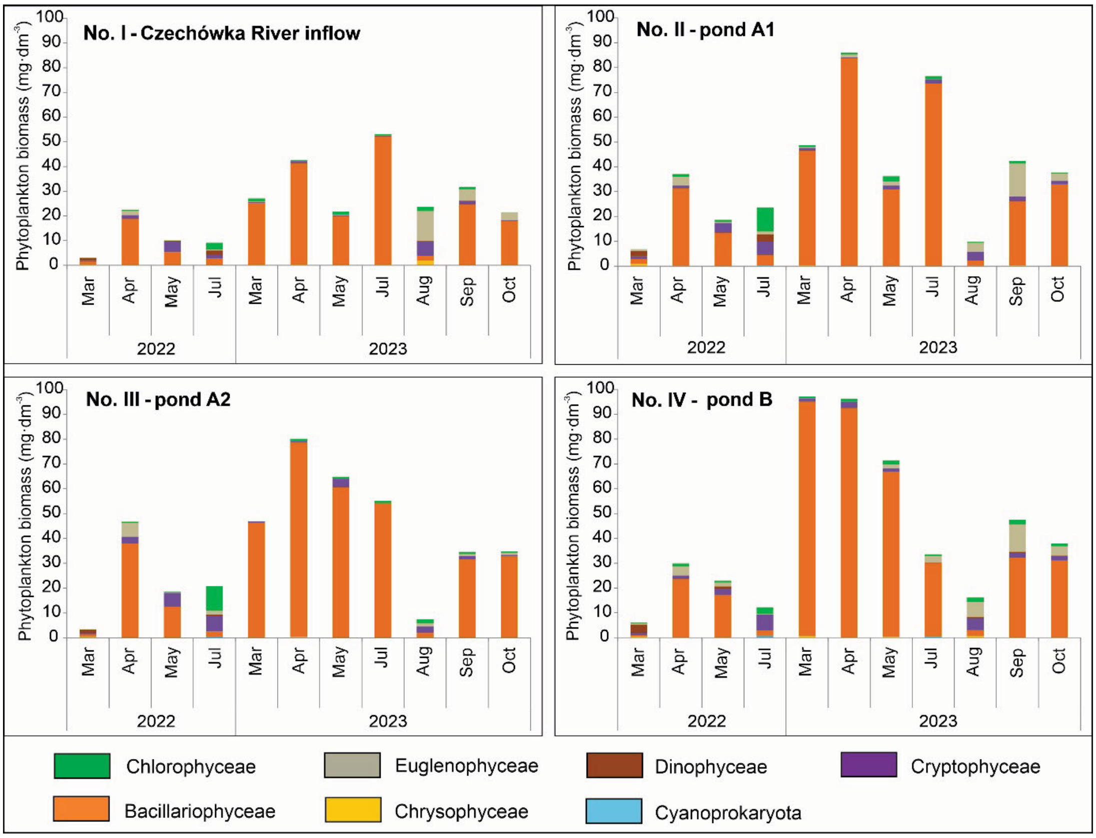


Fig. 7. Phytoplankton biomass (mg·dm^{-3}) in four study points – the Czechówka River (point I) and the ponds of the Botanical Garden of Maria Curie-Skłodowska University (points II, III and IV); source: own study

silicon-limited. Iron can also be an important factor regulating diatom growth (Brzezinski *et al*., 2011; Gao, Bowler and Kazamia, 2021), which has been confirmed for the dominant genus *Cyclotella* in our study (Lewandowska and Kosakowska, 2004). During periods of reduced diatom growth (July 2022), lower Fe values were recorded, both in the river and pond waters (Tab. S1). Moreover, the dominance of the eutrophic species *Cyclotella meneghiniana* for most months of our study may also have been influenced by management practices in the surrounding catchment area, such as agricultural fertilisation (Xie and Liu, 2001).

At low water temperatures (~4°C), the species composition of phytoplankton showed a dominance of *Chrysophyceae* in terms of abundance, while *Dinophyceae* dominated in biomass. During summer months (July 2022 and August 2023), when water temperatures were highest (>23°C) and concentrations of mineral forms of nitrogen and phosphorus decreased, there was a restructuring of the phytoplankton community structure in the studied waters. *Cryptophyceae* and *Chlorophyceae* became dominant in abundance, with additional presence of *Chrysophyceae* (August 2023 in both the river and pond B). In terms of biomass during summer in 2022, *Cryptophyceae* and *Chlorophyceae* played a decisive role, while *Euglenophyceae* dominated in August 2023 (in both the river and pond B). An increase in the proportion of *Euglenophyceae* in early autumn phytoplankton in nutrient-rich ponds was observed by Wołowski and Kowalska (2009). The observed seasonal changes in phytoplankton groups were typical for small, nutrient-rich water bodies (Kawecka and Eloranta, 1994; Messyasz and Jurgońska, 2003). In addition, the significant effect of water temperature on species structure and phytoplankton abundance was confirmed by statistical analysis ($p \leq 0.01$, Tab. S2).

In our studies, no significant contribution of cyanobacteria was observed in terms of both abundance and biomass of phytoplankton (constituting only a few percent of the total). Despite favourable conditions for their growth during the summer period – such as low nitrogen-to-phosphorus ratio, elevated water temperatures and windless weather – cyanobacteria remained at low levels. In July 2022, when TN:TP was ≤10, their abundance was similarly low compared to the following summer when TN:TP was significantly higher. Typical limiting factors for cyanobacteria growth, among others, water turbulence, rapid fluctuations in water temperature or intense grazing pressure from herbivorous fish, were not registered (Burchardt and Pawlik-Skowrońska, 2005). The rapid water exchange rate with the Czechówka River and the partial reduction of solar radiation reaching the water surface due to shading by trees overgrowing the banks of the ponds were among the significant factors limiting cyanobacterial growth in the studied ponds. Another possible limiting factor was the presence of macrophytic vegetation covering the pond bottom and edges. These acted as a buffer (Scheffer *et al*., 1993; Pęczuła, 2012; Widelska and Walczak, 2020), as such vegetation reduces sediment resuspension and absorbs nutrients in the water column.

CONCLUSIONS

Urban ponds are a key element of "blue-green infrastructure" in urban spatial planning. Their significant role in improving the quality of life for city residents necessitates a detailed understanding of the processes influencing their functioning, especially factors determining eutrophication.

The analysis of urban ponds in the Botanical Garden of Maria Curie-Skłodowska University in Lublin showed intensive phytoplankton growth during the years 2022–2023. The main cause identified for this phenomenon was the inflow of water rich in nutrients from the Czechówka River. Groundwater feeding into the studied ponds, despite high concentrations of mineral forms of nitrogen and phosphorus, had little significance in shaping water quality in the ponds. This is primarily because the ponds were isolated from groundwater by an impermeable polypropylene mat, with groundwater inflow occurring only through a low-yield spring.

To reduce the nutrient load entering the ponds, it would be advisable to implement biogeochemical barriers along the inflow from the river. Designing an effective sedimentation reservoir to reduce suspended solids is also a crucial aspect of pond modernisation. Moreover, the proper shaping of the shoreline and strategic planting of trees in the shoreline zone can significantly reduce water heating. It is important to maintain conditions that limit the growth of cyanobacteria by suitably shaping the macrophyte structure. This approach is not only beneficial for ecological reasons, but also enhances the aesthetic and recreational use of these water bodies.

SUPPLEMENTARY MATERIAL

Supplementary material to this article can be found online at https://www.jwld.pl/files/Supplementary_material_Ziolek.pdf

FUNDING

This work was supported by the National Science Centre Poland (grant numbers 2020/37/B/ST10/01994).

The project was funded from the state budget with funds granted by the Polish Minister of Education and Science within the framework of the "Excellent Science II" Programme. The paper was prepared for the "5th Crenology Conference. Springs – an underestimated phenomenon of nature".

CONFLICT OF INTERESTS

All authors declare that they have no conflict of interests.

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