



Response of a shallow water reservoir to anthropogenic stressors: implications for the water supply of Lake Velence, a major recreational lake in Hungary

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Abstract Reservoir pollution and eutrophication are pressing global issues threatening water supply and aquatic ecosystems. This study reconstructs multi-decadal environmental changes in the Pátka Reservoir (Central Hungary) using a 54-cm sediment core dated by $^{137}\text{Cs}/^{210}\text{Pb}$. Multi-proxy analyses included chironomid and pollen assemblages, chlorophyll degradation products (SPDU), major and trace elements and geochemical indicators (total organic carbon [TOC], total bound nitrogen [TbN], total sulphur [TS] and C/N ratio). Results revealed distinct ecological phases: the basal layers (1983–early 1990s)

showed dominance of semi-aquatic and opportunistic taxa, likely linked to the reservoir's drainage (1992) and refilling (1994). Between 1997 and 2015, macrophyte-associated taxa suggested mesotrophic–eutrophic conditions, while upper layers (2015–2022) were dominated by taxa indicating hypoxia and high trophic levels. Rising TOC, TbN, TS and SPDU [sedimentary pigment degradation unit] concentrations reflect intensifying eutrophic conditions, corroborated by monitoring data. Key drivers of ecological change were nutrient loading and water-level fluctuations in the 1990s. Given the reservoir's role in supplying Lake Velence and its ecological significance, reference conditions were defined and restoration strategies proposed, including angling regulation, ecological dredging and nutrient input control. Since March

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2024, the reservoir has remained drained due to poor water quality, highlighting urgent management needs.

Keywords Shallow lake ecosystem · Pátka Reservoir · Eutrophication · Human impact · Reference conditions · Chironomids

Introduction

Recent studies confirm that the combined effects of warming and accelerating anthropogenic stressors have caused changes in the ecological communities and water quality of lakes at different altitudes at an unprecedented rate (Adrian et al., 2009; Catalan et al., 2013; Moser et al., 2019), threatening the stable functioning of lakes (Dearing et al., 2014; Haliuc et al., 2020; Szabó et al., 2020; Magyari et al., 2024). Human activities present a formidable threat to freshwater ecosystems (Carpenter et al., 2011): the acceleration of industrialisation, agriculture, urbanisation and population growth during the twentieth century has increased nutrient influx, increased primary production, degraded water quality and reduced biodiversity worldwide (Ho et al., 2019). Anthropogenic eutrophication of lake ecosystems due to increased nutrient input is a widely known but largely unresolved environmental problem with only a few well-documented exceptions (e.g. Lake Constance, Ibrahim et al., 2022). Eutrophication, an increase in nutrient concentrations (nitrogen and phosphorus) and the alteration of physical and chemical water parameters (temperature, dissolved oxygen, pH, electrical conductivity) causes a reduction and homogenisation of aquatic biodiversity (Camargo et al., 2005) and often cyanobacterial blooms (Conley et al., 2009; Tarczyńska et al., 2001) which threaten water quality and the delivery of ecosystem services (Duffy et al., 2007; Feio et al., 2015; Ho et al., 2019). In general, human activities and the use of lakes for a variety of purposes increase with decreasing altitude, since lowland areas are preferred subjects to intense agricultural activities, urban development and population aggregation (Zhou et al., 2022). Thus, a lowland warm-temperature shallow lake is the perfect subject to follow the effects of anthropogenic impacts on a lake ecosystem.

Water-quality biomonitoring programs commonly use benthic macroinvertebrates as bioindicators of environmental pollution (Valdovinos and Figueroa, 2000; Callisto et al., 2001). Their communities have also frequently been used in the assessment of human impacts on freshwater ecosystems (Rosenberg & Resh, 1993; Fleituch et al., 2002). The largest group of macroinvertebrates is the non-biting midges (Chironomidae) that play an essential role in aquatic ecosystems. Their larval head capsules preserve well in lake sediments, and in most cases they are the most abundant macrobenthic animals in freshwater environments (Luoto & Nevalainen, 2015). The narrow ecological tolerances of species make them ideal for lake-water-quality monitoring (Walker, 2001; Holmes, 2014). In alpine and boreal lakes with cool summer and limited nutrient availability, the community composition is usually driven by July or summer mean temperature (Heiri et al., 2011; Luoto & Nevalainen, 2015). However, in lowland shallow lakes, other environmental factors associated with human impact (e.g. salinity, trophic level, pH, aquatic vegetation changes, fish predation) usually exceed the role of July mean temperature (Langdon et al., 2006; Nevalainen & Luoto, 2012), which makes chironomids an appropriate tool to reconstruct the water-quality status of lakes in the past (Luoto, 2011).

Lake Velence, one of Europe's westernmost steppe lakes, is one of the most important standing waters in the Danube basin. It is a shallow (average depth 1.5 m) and heavily infilled soda lake, covering an area of 25 km² (Baják et al., 2022). The southwestern part of the lake is a nature conservation area with a wide reed belt. The central and northeastern parts are open water (Reskóné, 1999; Reskóné et al., 2001) (Fig. 1). The lake serves multiple purposes and provides a range of ecosystem services, including provisioning services (e.g. drinking water, angling), regulating and supporting services (e.g. nature conservation and ecosystem complexity), as well as cultural services (e.g. recreation, sports, tourism, natural beauty and inspiration) (Kovács-Hostyánszki et al., 2022; Tanács et al., 2021). A natural aspect of its hydrological regime includes the periodic drying of certain areas and considerable fluctuations in water depth in others. Notably, the lake has recently reached record-low water levels (Riezing, 2023).

To maintain the required water level and prevent its extreme fluctuation, two reservoirs were established

along the inlet, Császár Creek (Baják et al., 2022), which transfers water from 67% of the catchment area. By operating the Zámoly and Pátka Reservoirs, the minimum water depth during the bathing season remains above 120 cm with 90% probability at the city of Agárd (Significant water management issues, 2020). The organic matter and nutrients from Császár Creek have a negative impact on the water quality of Lake Velence Nature Reserve, which is a reedbed zone in the western subbasin resulting in increased phosphorus concentrations and chlorophyll-a content in the water, which promote algal blooms (Ács et al., 1994, 2005). Formerly, the negative effects of local water-quality deterioration were isolated (e.g. mass fish mortality due to temporary oxygen depletion), but lately the effects spread without control throughout the whole lake (e.g. the mass proliferation of the toxic blue-green alga *Microcystis aeruginosa* (Kützling) Kützling, 1846 within the plankton (Ács et al., 1994; Reskóné and Törökné, 2000)). These facts highlight the importance of water-quality improvement in Lake Velence. Although lakes can naturally be eutrophic in certain geological settings, eutrophication, the process of increasing nutrient concentrations, is primarily a consequence of human activities that increase nutrient loading beyond natural levels (Engstrom & Fritz, 1990; Ansari et al., 2011). For these reasons, the maintenance of good water quality in reservoirs is crucial due to their importance for water management, recreation, ecology and related purposes, which is the case in Lake Velence.

In this paper, we focus on the Pátka Reservoir, which has a significant impact on the water quality of Lake Velence through its water discharge. We analyse a gravity sediment core taken from it, covering approximately the last 40 years. Although monitoring studies are abundant in reservoirs (Moreno & Callisto, 2006; Gomes et al., 2018; Zinchenko & Shitikov, 2021), the application of sediment chemistry (Baran et al., 2019) and fossil faunistic and floristic analyses are less common (Clark, 1986; Lucas et al., 2015a; Brooks et al., 2016). Therefore, our approach is relatively new in this field. In this study, we apply paleoecological (chironomid, pollen) and geochemical methods (sediment chlorophyll (SPDU), TOC, TS, TbN, C/N ratio, major and trace element concentration measurements, grain size analysis) along with radiometric dating to obtain information on the timing and magnitude of human activities, the process of eutrophication and its consequences

for the lake's chironomid fauna (Cohen, 2003; Taylor et al., 2013).

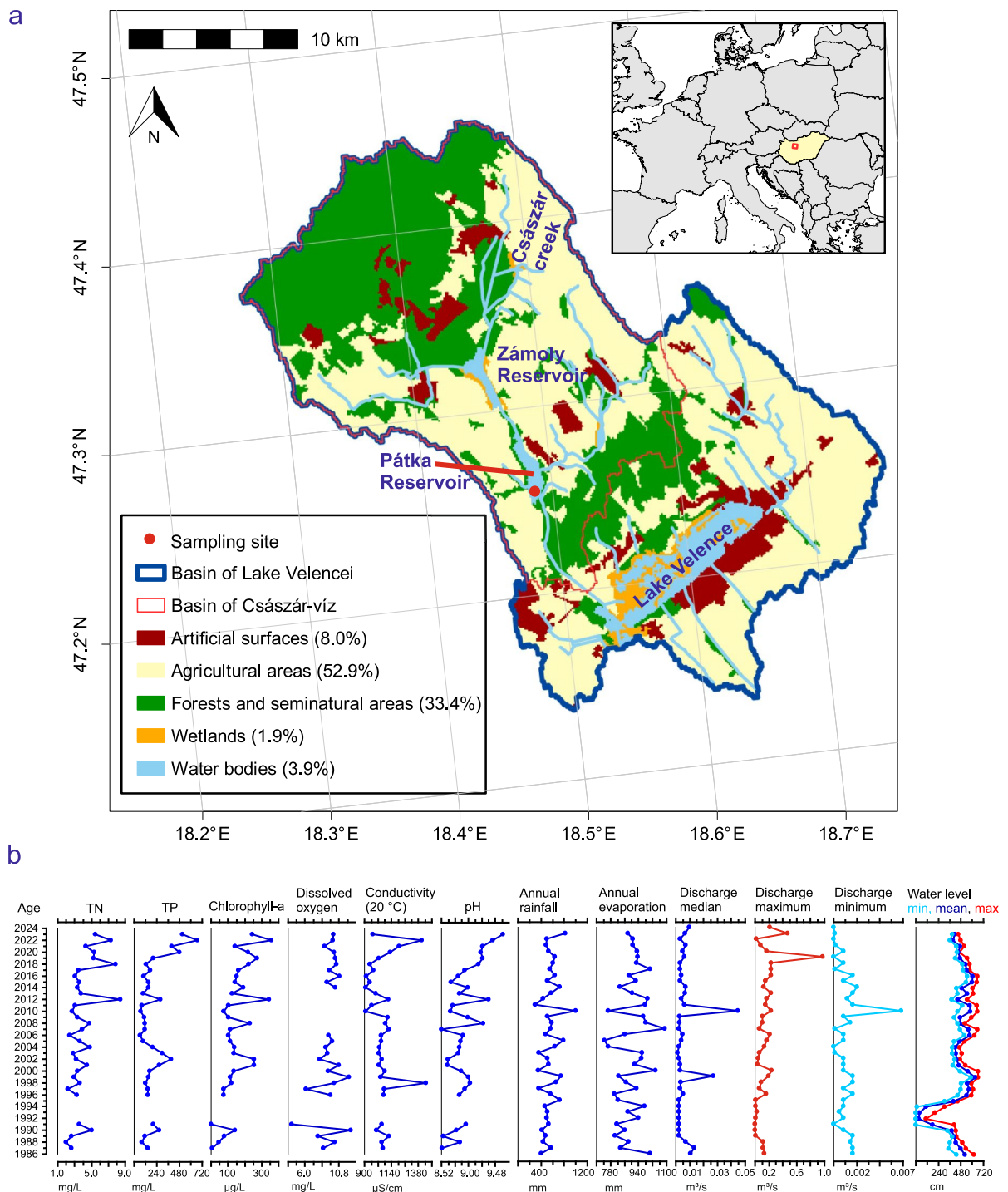
The relevance of our research is further supported by the fact that rising temperatures and changing precipitation patterns have made water security a critical issue in many regions worldwide. In the Carpathian Basin—particularly in the Great Hungarian Plain—alterations in the two primary climatic variables (precipitation and temperature), coupled with the intensification of human activities, have triggered significant, long-term transformations in the landscape over recent decades (Ladányi et al., 2009). The warming trend and decreasing precipitation are contributing to groundwater depletion—further exacerbated by irrigation and the construction of water reservoirs in response to reduced rainfall—which in turn leads to changes in both soil conditions and landscape structure (Rakonczay, 2011). The fluctuations in water level and quality observed in Lake Velence reflect this pattern, highlighting the urgent need for effective mitigation measures.

Our working hypothesis is that the studied reservoir has gone through substantial ecosystem reorganisation in response to increasing nutrients since its establishment in 1974, due to multiple factors that contributed to a rapid faunal turnover. Accordingly, our aim is to assess the ecological development of the reservoir through a multi-proxy approach combining faunal (Chironomid), geochemical and pollen analyses together with historical water-quality data. This integrated assessment allows us to identify the faunal assemblage representing the good ecological status of the water body and to define restoration targets and management actions necessary to achieve a sustainable and healthy state of the reservoir (Dearing et al., 2014).

Materials and methods

Study site and water-quality changes since 1987

Pátka Reservoir (Fig. 1; N 47.2568 °, E 18.4902 °) was established in 1974 by the damming of the Császár Creek. It provides water for Lake Velence together with the Zámoly Reservoir. Located at an altitude of 137 m a.s.l., the reservoir covers an area of 2.9 km² and has an average water depth of 2.5 m. The total catchment area is 351 km². Its inflowing watercourses are Császár Creek and Rovájka Creek. According to information available in the annual



water budget reports on the official website of the Central Transdanubian Water Directorate (Water budget of Lake Velence 1986–2024), the catchment is characterised by a sub-Mediterranean climate

with approximately 570 mm annual precipitation and 11.4 °C average annual temperature. The catchment area experiences a mean January temperature of -1.1 °C and a mean July temperature of 21.1 °C,

Fig. 1 (a) Location of the Pátka Reservoir and the sampling site in the catchment area of Lake Velence in Europe, Hungary showing Corine 2018 land cover categories (Feranec et al., 2016) (b) Mean summer water-quality variables (total nitrogen [TN], total phosphorus [TP], chlorophyll-a concentration, dissolved oxygen, conductivity and pH), measured from the water of Pátka Reservoir by KDTVIZIG (Central Transdanubian Water Directorate); annual rainfall and annual evaporation data in the watershed of Császáz Creek; annual median, maximum and minimum discharge near Csákvár (N 47.389311 ° E 18.480808 °) between 1986 and 2024; annual minimum, mean and maximum water level in the Pátka Reservoir between 1986 and 2023 based on data from Central Transdanubian Water Directorate (b)

indicating a marked seasonal variation. The Pátka Reservoir is part of the National Ecological Network, within which it serves as an ecological corridor (Kovács, 2014).

Until recently it has been an angling paradise, but its original water supplying function for Lake Velence became limited in recent years due to water quality problems. Earlier, the total drainage of the reservoir was necessary in the early 1990s due to summer drought threatening Lake Velence (Water budget of Lake Velence 1986–2024). In its recent state, the reservoir was suitable for recharging Lake Velence only in early spring or late autumn and winter (Water budget of Lake Velence 1986–2024), due to its high chlorophyll-a content, which exceeds the hypertrophic limit and threatened the water quality of the recipient Lake Velence. Water quality in the reservoir has been detected to be poor since the 1990s, with annual chlorophyll peaks of around 500 mg/m³ recorded in the 2000s (János Kóbor pers. comm.). Although the general mechanisms of eutrophication are well known (Dillon & Rigler, 1974), the local causes of water-quality deterioration in the Pátka Reservoir have not yet been scientifically investigated. It is assumed that the main causes are the accumulation of sediment due to lack of dredging and the large amount of bait thrown into the water by anglers, which add a lot of extra nutrient (Significant water management issues, 2020). The decay of deposited organic matter causes oxygen deficiency that has recently resulted in mass fish mortality, leading to the decision to drain the reservoir in March 2024 and remove the fish (Vaskor, 2024). The results of a study carried out over five days at five measuring points showed that the quality of the water discharged into Lake

Velence via the Császáz Creek did not improve significantly over the 10 km length of the stream, and the chlorophyll-a content of the water entering the lake also did not change significantly (Szily, 2021), which made it clear that the water quality of the reservoir has to be improved for the safe water supply of Lake Velence.

Regarding water-quality monitoring data (1987–2023) collected by KDTVIZIG (Central Transdanubian Water Directorate), as Fig. 1b shows, TN and TP both increase in the most recent sediments. TP displayed a rapid increase after 2018, characterised by the highest value (670 mg/L) and a secondary peak in 2002 (394 mg/L). TN showed greater fluctuations, with peaks in 2012 (8.4 mg/L) and 2018 (7.9 mg/L). Chlorophyll-a showed a strongly fluctuating yet increasing trend. The summer of 2022 was characterised by the highest measured values (mean: 356.7 µg/L), but the summer of 2012 displayed similarly high values (343.5 µg/L). Dissolved oxygen concentrations showed fluctuations but remained relatively stable throughout the studied period. The lowest values occurred in 1991 (4.2 mg/L) and 1997 (6.2 mg/L). Conductivity was stable between 1987 and 2009 (~1082 µS/cm), followed by relatively low values between 2010 and 2018 (965 µS/cm), then a rapid rise to 1408 µS/cm and a drop in 2023 (974 µS/cm). pH showed an upward trend with strong fluctuations. The highest values were measured after 2018 (> 9), but in 1998, 2008 and 2012 other peaks were detected (Fig. 1b).

Regarding the meteorological parameters, annual rainfall in the Lake Velence watershed averaged 552 mm exceeding 700 mm in 6 years (1995, 1999, 2005, 2010, 2014 and 2023) for the period 1986–2023. The lowest annual rainfall occurred in 2011 (< 300 mm). Average evaporation for the same period was 902 mm, exceeding 1000 mm in 1986, 2000, 2007 and 2017, whereas 2004, 2005 and 2010 had the lowest evaporation (< 800 mm). The median daily discharge at Császáz Creek exhibited two notable peaks in 1999 and 2010, both coinciding with particularly wet years. The mean, minimum and maximum water levels of the reservoir displayed remarkable changes between 1986 and 2023. The drainage of the reservoir is clearly visible in the data between 1991 and 1994. The mean water level was 457 cm between 1996 and 2016 with an increasing trend

between 2006 and 2016, followed by a continuous decrease and total drainage in March 2024 (Fig. 1b).

Sediment sampling

The 54-cm short sediment core from Pátka Reservoir was collected in September 2022 from the southern part of the lake (N 47.2568 °, E 18.4902 °; Fig. 1) using a gravity corer with a 7-cm diameter. At the core location, water depth was 1.1 m. The core (Pátka-t-2) was sliced into 1-cm sections in the field and stored in self-sealing plastic bags at 4 °C until further analyses.

$^{210}\text{Pb}/^{137}\text{Cs}$ dating

Dating by ^{210}Pb and ^{137}Cs was carried out on the sediment core Pátka-t-2. Sub-samples were analysed for ^{210}Pb , ^{226}Ra and ^{137}Cs by direct gamma assay at the Liverpool University Environmental Radioactivity Laboratory, using Ortec HPGe GWL series well-type coaxial low-background intrinsic germanium detectors (Appleby et al., 1986). ^{210}Pb was determined via its gamma emissions at 46.5 keV, and ^{226}Ra by the 295 keV and 352 keV γ -rays emitted by its daughter radionuclide ^{214}Pb following 3 weeks of storage in sealed containers to allow radioactive equilibration. Supported ^{210}Pb activity was assumed to be equal to the measured ^{226}Ra activity, and unsupported ^{210}Pb activity was calculated by subtracting supported ^{210}Pb from the measured total ^{210}Pb activity. ^{137}Cs was measured by its emissions at 662 keV. The absolute efficiencies of the detectors were determined using calibrated sources and sediment samples of known activity. Corrections were made for the effect of self-absorption of low-energy γ -rays within the sample (Appleby et al., 1992). For age-depth modelling, the constant rate of supply (CRS) model was used (Appleby & Oldfield, 1978).

To account for uncertainties in the age-depth model, it should be noted that sediment accumulation rates in the upper 30 cm of the core are variable, as indicated by the irregular decline of unsupported ^{210}Pb . Although the ^{137}Cs peak provides a well-defined time marker corresponding to the 1986 Chernobyl accident, interpolated ages between dated levels are approximate. Consequently, the

assigned ages should be interpreted as decadal-scale estimates rather than precise annual dates.

Organic content and sediment chemistry (major and trace element concentrations)

Loss-on-ignition (LOI) analyses were performed on 1-cm³ subsamples taken contiguously at 2-cm intervals. Samples were combusted at 550 °C for 4 h (Heiri et al., 2001). Geochemical element analyses were done at 2-cm resolution. For the major and trace element analyses 0.15–0.25-g samples were digested with a mixture of 2.5 ml 65% HNO_3 (Merck, Suprapur) and 7.5 ml 30% HCl (Merck, Suprapur). Following microwave digestion (Speedwave Entry, Berghof, Germany), the solutions were diluted with ultrapure water. After the element-specific dilution, CsCl was added as matrix modifier and elemental analyses were performed with a microwave plasma atomic emission spectrometer (MP-AES Model 4210, Agilent Technologies, USA). The atomic emission in the nitrogen plasma provides better detection limits and wider linear dynamic range compared with the flame atomic absorption technique. The concentration of Al, Ba, Ca, Cr, Cu, Fe, K, Li, Mg, Mn, Na, Pb, Si, Sr and Zn was quantified. The measurements took place at the Environmental Science Department at the Sapientia Hungarian University of Transylvania in Cluj-Napoca, Romania.

TbN, TS and TOC measurements

A sub-sample of 1 cm³ was taken from the sediment core at 2-cm intervals to determine total carbon, total sulphur (TS), inorganic carbon and total bound nitrogen (TbN) content with an elemental analyser (Elementar varioMacroCube) using the Dumas method (Dumas, 1831). Inorganic carbon was determined using an infrared analyser combined with a furnace (Shimadzu TOC-L SSM5000). Total organic carbon (TOC) was calculated as the difference between total carbon and inorganic carbon. The measurements took place at the Geographical Institute of the HUN-REN Research Centre for Astronomy and Earth Sciences in Budapest, Hungary.

Sedimentary pigment degradation unit (SPDU)

SPDU, or Sedimentary Pigment Degradation Units, is a method to measure the amount of chlorophyll and its decomposition products in sediment samples, the sedimentary pigment decomposition products per gram dry weight. Since it is an arbitrary unit, we used 1-cm³ wet sediment taken at 1-cm interval which was topped up with 10 ml acetone to bring chlorophyll derivatives into solution. Absorbance values were measured with a Jenway 6705 UV/Vis spectrophotometer in the wavelength range of 665 and 750 nm from the supernatant. The weight of the solid phase was determined after the evaporation of acetone and drying to constant weight at 105 °C. SPDU content was calculated using the formula of Vallen-tyne (1955) and expressed relative to 1 g dry weight (SPDU DM⁻¹).

Grain size distribution

Physical pre-treatment, ultrasonic treatment at 130 W and 28 kHz (McCave & Syvitski, 2010), was used prior to analysis to promote aggregate dispersion, while chemical pre-treatment was used to remove adhesives (organic matter and CaCO₃) (Gee & Bauder, 2018). Chemical pre-treatment was performed using sodium pyrophosphate supplemented with hydrogen peroxide treatment to remove organic matter. Grain size distribution was determined using a laser diffraction grain size analyser (Horiba LA-950V2).

Chironomid analysis

The volume of sediment processed for the chironomid analysis varied between 4 and 10 cm³ depending on the number of head capsules in 1-cm³ wet sediment. The analysis was performed at 5-cm resolution using 2-cm-thick sediment sections. Due to the very low head capsule concentration, the 1–2-cm and 4–5-cm samples were merged into a sample representing the core section 1–5 cm, whereas the 49–50-cm and 53–54-cm samples were merged into the 49–54-cm sample. The recommended minimum of 45–50 head capsules (Heiri & Lotter, 2010) is still not achieved at the bottom and top of the core, at 1–5-cm and 49–54-cm samples. Head capsules were sorted from Bogorov's counting trays (Gannon,

1971) under a stereomicroscope at ×40 magnification after heating in 10% KOH solution (15 min, 85 °C) and sieving at 100 micron. Head capsules were mounted on microscope slides in Euparal® mounting medium for identification. For the identification to genus or morphotype level, we used the works of Brooks et al. (2007) and Andersen et al. (2013) and an Olympus CX41 microscope at ×100–400 magnification. The identified specimens were classified into feeding guilds according to their feeding habits following the description of Heino (2008) and Luoto and Nevalainen (2015). Ecological groups based on trophic status were defined based on the information in Klink and Moller Pillot (2003) and Brooks et al. (2007).

Pollen analysis

Pollen analysis was done at 2-cm resolution to examine vegetation changes. 1-cm³ wet subsamples were prepared using standard methods (HCl, NaOH, HF, acetolysis and sieving between 10 and 125 µm) (Bennett & Willis, 2001). Pollen and microcharcoal (> 10 µm) particles were counted and identified under an Olympus BX43 light microscope at ×400 and ×1000 magnification. At least 500 terrestrial pollen grains were counted on each slide. For pollen identification, atlases of Reille (1992, 1995, 1998) and Moore et al. (1991) were used.

Interpretation of KDTVIZIG water-quality monitoring data and meteorological parameters

The KDTVIZIG operates a monitoring system to provide up-to-date information on water quantity and quality under their management. We interpreted the results of selected regular water chemistry tests (TN, TP, chlorophyll-a, dissolved oxygen, conductivity, pH) for the summer period (June, July, August), water discharge, water-level data and meteorological parameters since 1987 from the Pátka Reservoir. The dataset lacked measurements regarding water-quality parameters from the years 1992, 1993, 1994 and 1995 since the reservoir was drained at that time.

Statistical methods

The chironomid and pollen relative frequency diagram and geochemical concentration diagrams were

plotted using the C2 and Psimpoll 4.27 software (Juggins, 2003; Bennett, 2007). Stratigraphically constrained hierarchical cluster analysis CONISS (Grimm, 1987) was used to identify chironomid assemblage zones in Psimpoll 4.27, whereas optimal splitting by information content was used to determine pollen and geochemical zones taking into account pollen types with > 5% relative abundance and all geochemical element concentrations, respectively. Input data for chironomid and pollen zonation were ratios, whereas that for geochemical zonation were concentrations. The statistical significance of zone boundaries was tested with the broken stick model (Bennett, 1996). Due to the assumed linear relationship between our variables, we chose principal component analysis (PCA) to analyse trends in the proxy records. The chironomid and pollen relative frequency data were square-root transformed (Legendre & Gallagher, 2001), while geochemical element concentrations were z-transformed (Hair et al., 2019) prior to PCA. PCA was run on covariance matrix in the case of chironomids (including taxa with relative abundance > 5%) and pollen (including taxa with relative abundance > 2%). In the case of the geochemical element PCA, we used correlation matrix. Chironomid and palynological richness were determined in Psimpoll 4.27 using rarefaction analysis (Birks & Line, 1992). Pairwise correlation between the concentration values of geochemical elements was calculated using Spearman's rank correlation (Spearman, 1904). Only relationships with $p < 0.05$ were considered significant. Redundancy data analysis (RDA) was used to explore the relationship between changes in the chironomid assemblages and other environmental variables. This analysis was chosen because the gradient length of the first axis was less than four SD units (Legendre & Legendre, 2012). RDA used the PC1 and PC2 axes of the geochemical records, PC1 and PC2 axes of the pollen record, LOI, SPDU DM^{-1} , TbN, TOC, TS, the proportion of sand, silt, clay and the element concentrations as explanatory variables.

Results

$^{210}\text{Pb}/^{137}\text{Cs}$ dating

The ^{210}Pb results indicate that the 54-cm core represents sediment deposited over the last ~40 years. Total ^{210}Pb activity in the deepest sample (54 cm)

exceeds that of supported ^{226}Ra , showing that the core does not cover the full ^{210}Pb timespan (typically around 100 years). In sediments below 30 cm, unsupported ^{210}Pb declines with depth, while above 30 cm the decline becomes more variable. A well-resolved ^{137}Cs activity peak occurs at 52 cm (7164 Bq m^{-2} , or $16,368 \text{ Bq m}^{-2}$ when decay-corrected to 1986). Dates were calculated using the CRS model with the ^{137}Cs peak as a reference point. Based on these calculations, sediment accumulation rates fall around 1.28 cm y^{-1} from the mid-1980s to around 2010, and around $0.69 \text{ g cm}^{-2} \text{ y}^{-1}$ during the past decade. Volumetric accumulation in the uppermost layers reaches approximately 2 cm y^{-1} (Fig. 2).

Grain size distribution, organic matter and sediment chemistry

The sediment is characterised by relatively low organic content (10.3%) between 52 and 26.5 cm (1983–2008) followed by a steep increase between 26.5 and 6.5 cm (2008–2019) (Fig. 3; ESM Fig. 1) and a subsequent decline in 2020. The upper 5.5 cm was characterised by a gradual increase reaching a maximum in 2022.

Cyclically varying amount of sand, silt and clay characterises the sediment core between 52 and 28.5 cm (1983–2007). After 2007, sand content increased significantly (from 1.4 to 60.6%) until 2021.

Out of the 15 elements measured by the spectrometer (ESM Fig. 1), we discuss Fe, Al, Mg, Si and Ca that were the most abundant (450 to $180,700 \text{ mg/kg}$). In addition, we focus on heavy metals, Zn, Pb and Cu. The trends in the concentration of these elements are displayed in Fig. 3. Four significant zones were identified in the geochemical record based on optimal splitting by information content, the latter two were subdivided in two sub-zones visually (Fig. 3; ESM Fig. 1).

The first two axes of geochemical element PCA (Fig. 3; ESM Fig. 2) explain 68.4% of the total variance (PC1: 57%; PC2: 21.4%). Almost all loadings along the first axis had negative values, even the most positive loadings are around 0, these are associated with Si, Zn and Pb. The negative direction along PC1 is associated with Na, Cu and Sr. Along the PC1 component, the more stable deeper layers are dominated by carbonate sediments (Ca, Ba, Sr) from the bedrock or biogenic carbonate precipitation, whereas in the

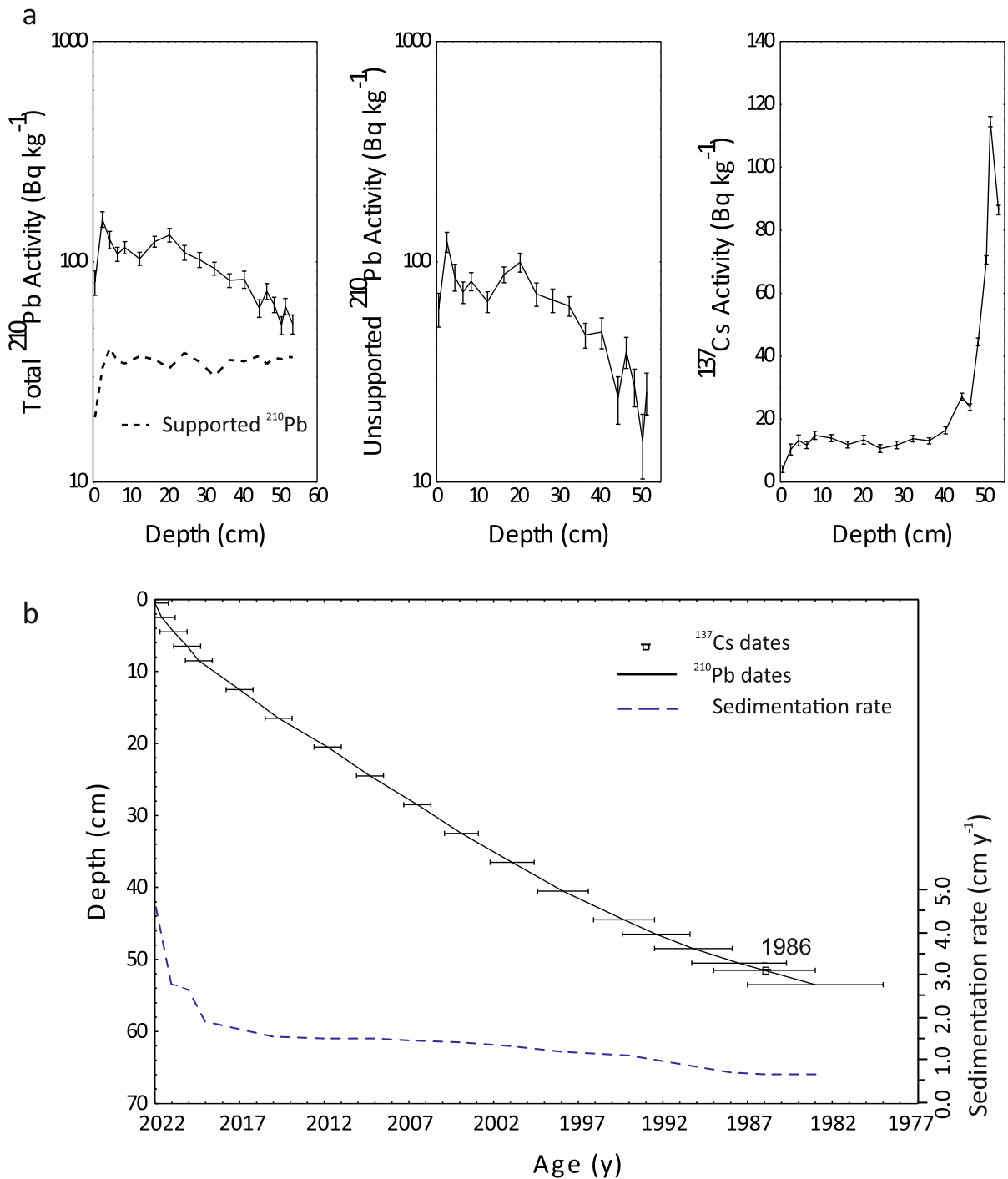


Fig. 2 (a) Fallout radionuclides in the Pátka Reservoir sediment core Pátka-t-2 showing total and supported ²¹⁰Pb, unsupported ²¹⁰Pb and ¹³⁷Cs concentrations versus depth (b) Radiometric chronology of the Pátka Reservoir sediment core

Pátka-t-2 showing the 1986 depth suggested by the ¹³⁷Cs record, and the ²¹⁰Pb dates and sedimentation rates calculated from the CRS model using the ¹³⁷Cs date as a reference point

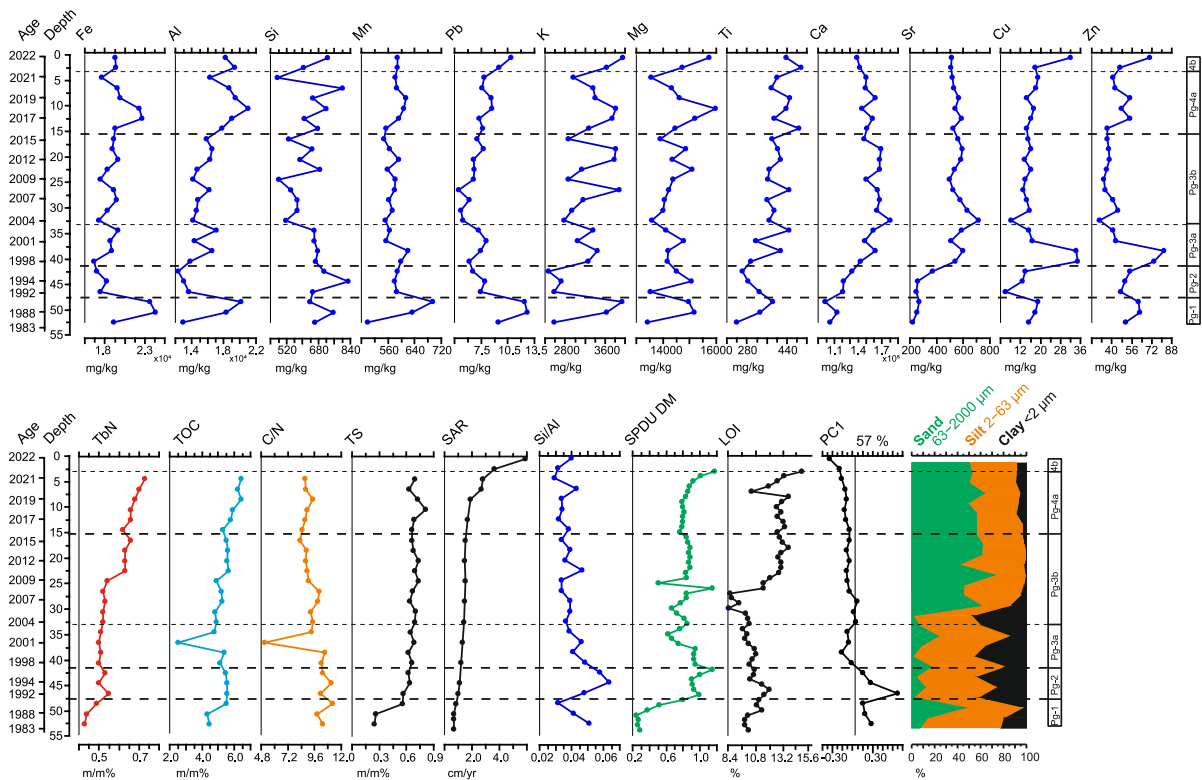


Fig. 3 Selected geochemical element concentrations, total bound nitrogen content (TbN), total organic carbon content (TOC), carbon/nitrogen ratio (C/N), total sulphur content (TS), sediment accumulation rate (SAR), chlorophyll derivatives (SPDU DM^{-1}) relative frequency change (relative to dry

weight), organic matter content (LOI), PC1 stratigraphic plot with the percentage of the total variance the axis explains and the result of the particle size analysis from the sediment of the Pátkai-t-2 gravity core, Pátka reservoir, Hungary

upper layers (post-2016) anthropogenic heavy metal accumulation is observed. The highly negative loading of Na is associated with the period 1992–1994 of the drained reservoir. The most positive values along the PC2 axis are associated with Pb, Cu and Zn, while the negative direction is associated with Ca and Sr again, differentiated the clayey and heavy metal contaminated layers from the deeper carbonate rich layers.

In Zone Pg-1 (52.5–47.5 cm, 1983–1991), Fe, Al, K, Mg, Si, Cr, Zn, Cu, Pb and Ti show increasing trends and high values, accompanied by increasing sand content (max. 46.5%). The ratio of Si/Al is elevated.

In Zone Pg-2 (47.5–41.5 cm, 1991–1997), a decreasing trend is present in Fe, Al and K. Mg changes in the opposite direction along with Si compared to Fe and Al. TbN, TS, TOC and SPDU DM^{-1} were stable in this zone, and the proportion of clay

increased (mean: 37.5%). PC1 sample scores shift towards more positive values.

In Pg-3a (41.5–33.5 cm, 1997–2004 (± 3 years)), minerogenic elements (Fe, Al, K, Ti) increase together with Ca. Zn, Pb and Cu increase around 1998–2000. Correlations with clastic elements remain weak (Al–Zn: 0.17; Al–Cu: 0.47) (ESM Table 1). TOC decreases abruptly, as does SPDU DM^{-1} . Grain size displays cyclic alternation between more sandy and more clay-rich layers. Pg-3b (33.5–15.5 cm, 2004–2015 (± 2 years)) was characterised by the fluctuation of detrital elements with a slight overall increase. TbN increases in 2009–2010, TOC shows a slight increase and LOI increases after 2008. SPDU DM^{-1} increases abruptly in 2008, then decreases and later stabilises. Sand content increases markedly in 2007 (mean 55%) and remains high together with Si

concentrations until the present (Fig. 3). Ca reaches its highest values in this zone.

In zone Pg-4 (15.5–0 cm, 2015–2022 (± 2 years)), TbN, TOC, LOI and SPDU DM^{-1} continue to increase, while Ca shows a slight decrease. Organic matter decreases abruptly in 2020. In Pg-4a (15.5–3.5 cm, 2015–2021), Fe, Al, Mg, Si and K increase until 2017–2018 (Fig. 3; ESM Fig. 1). Pg-4b (3.5–0 cm, 2021–2022) is characterised by a further increase in Al, Cr, Li, Pb, K, Mg, Ti, Cu and Zn.

Chironomid stratigraphy

A total of 29 chironomid taxa were identified from the sediment core (Fig. 4a). Three main zones were distinguished based on CONISS cluster analysis (Fig. 4).

Pch-1 (42–54 cm, 1983–1997 (± 6 years)) is characterised by low taxon numbers (15) and low chironomid concentrations ($6.5 \text{ head capsules cm}^{-3}$) (Fig. 4b). The assemblage is dominated by *Chironomus plumosus*-type (25.8%) and semi-aquatic

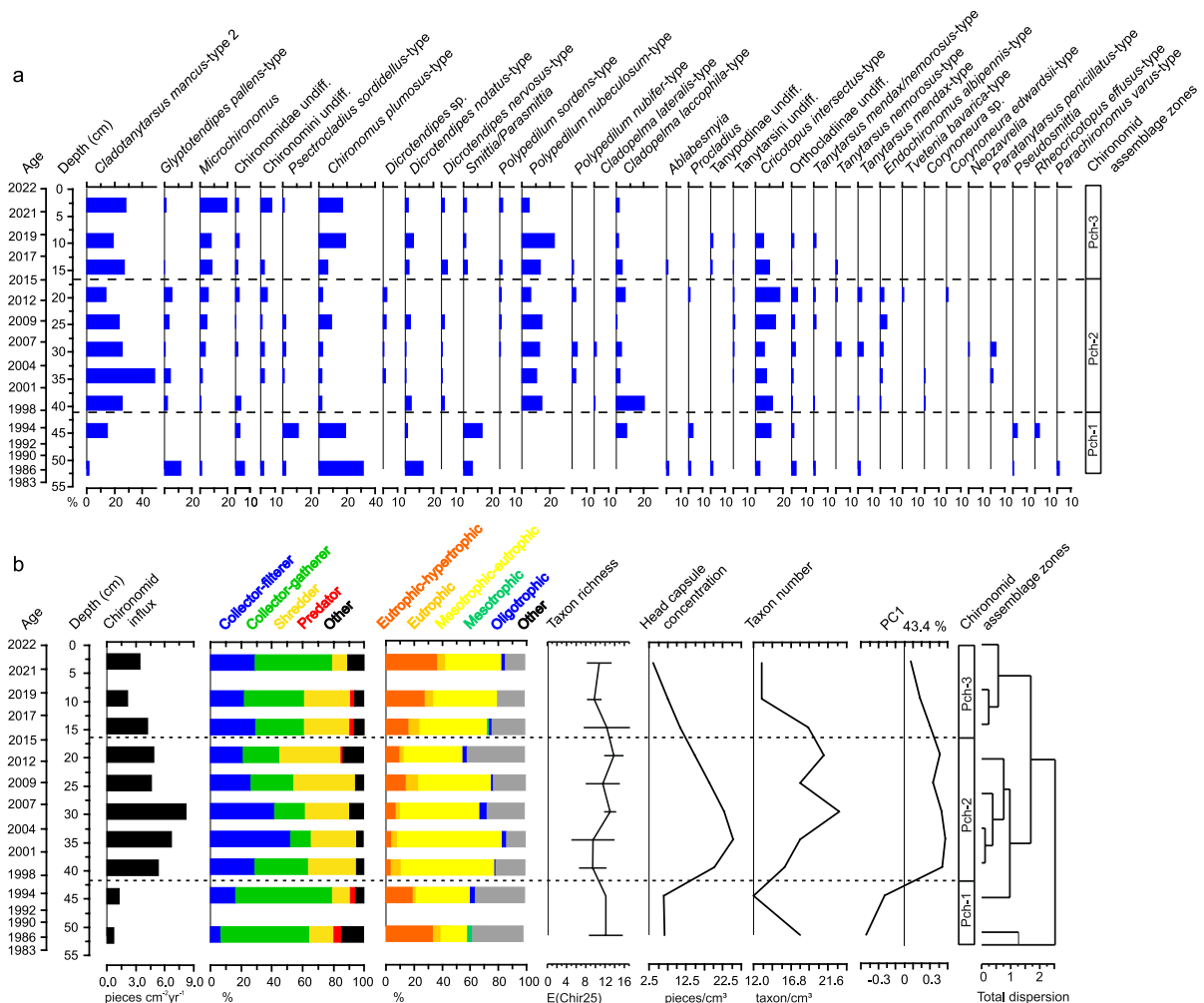


Fig. 4 (a) Relative chironomid frequency diagram showing the identified species from the sediment of Pátka reservoir, core Pátka-t-2 (b) three chironomid assemblage zones were distinguished using stratigraphically constrained cluster analysis (CONISS, Grimm 1987); chironomid influx, chironomid feeding guilds (blue: collector-filterer, green: collector-gath-

erer, red: predator, yellow: shredder, black: others), relative frequencies of trophic tolerance classes, taxon diversity (taxon richness), HC (chironomid head capsule) concentration, the number of chironomid taxa and PC1 sample scores with the percentage of the total variance the axis explains along age and depth from Pátka-t-2

Smittia/Parasmittia, with the presence of vegetation-associated *Cricotopus intersectus*-type and *Psectrocladius sordidellus*-type. Rheophilic or running-water-associated taxa, including *Rheocricotopus effusus*-type (3.9%), *Dicrotendipes notatus*-type (7.6%) and *Parachironomus varus*-type (1.9%), are also present (Fig. 4a). Taxon richness remains low and stable (12). Chironomid influx reaches the lowest values in the core (mean 1.1 pieces cm⁻² yr⁻¹). Collector-gatherers dominate the functional feeding composition (60.1%) (Fig. 4b).

Pch-2 zone (17–42 cm, 1997–2015 ($\pm$ 2 years)) displays a more diverse fauna, with higher proportions of *Cladotanytarsus mancus*-type 2 (28%), *Cricotopus intersectus*-type (12.5%) and *Polypedilum nubeculosum*-type (12%) (Fig. 4a). Head capsule concentration (25.3 head capsules cm⁻³), species number (23) and chironomid influx (8.3 pieces cm⁻² yr⁻¹) reach their maximum between 29 and 35 cm (2003–2007; Fig. 4b). Several warm-water taxa with low relative abundances appear (e.g. *Paratanytarsus penicillatus*-type, *Endochironomus albipennis*-type, *Tanytarsus mendax*-type). Taxon richness increases again, reaching a maximum of 13.7 at 20 cm (2011). Functional groups shift from collector-gatherers towards collector-filterers until 35 cm and then show an increase in the proportion of shredders (maximum 40.4%) and collector-gatherers (maximum 27.8%) (Fig. 4b).

In zone Pch-3 (0–17 cm, 2015–2022 ($\pm$ 2 years)) the most notable change is the increasing relative abundance of *Chironomus plumosus*-type (up to 14.5%). *Glyptotendipes pallens*-type decreases (to 0.7%), as does *Cricotopus intersectus*-type. The relative abundance of *Dicrotendipes* increases (6.7%). Semi-aquatic *Smittia/Parasmittia* (2.7%) appears again. *Microchironomus* increases to 12.6%. Several taxa disappear from this section, including *Endochironomus albipennis*-type, *Cladopelma lateralis*-type, *Procladius* and *Tanytarsus mendax*-type. Taxon number (13), taxon richness (10.1), head capsule concentration (5.6 pieces cm⁻³) and chironomid influx (3.3 pieces cm⁻² yr⁻¹) all decrease compared to the previous zone (Fig. 4b). In the uppermost sample, the proportion of collector-gatherers reaches 50%.

The first two axes of chironomid PCA explain 61.7% of the variance (PC1: 43.4%; PC2: 18.3%). Positive PC1 values are associated with meso-eutrophic and vegetation-associated taxa (*Polypedilum nubeculosum*-type, *Cladotanytarsus mancus*-type

2), while negative scores correspond to opportunistic or semiterrestrial forms (*Chironomus plumosus*-type, *Smittia/Parasmittia*) (ESM Fig. 3).

Pollen stratigraphy

Six pollen assemblage zones were identified using optimal splitting by information content, but the two lowermost zones were merged due to overall similarity. The zones are hereafter referred to as Pp-1 to Pp-5 (Fig. 5).

The first two axes of the pollen PCA explain 62.2% of the total variance. High positive loadings of arboreal taxa (*Pinus* L., *Salix* L., *Fagus sylvatica* L., *Alnus* Mill., *Betula* L.) occur on PC1 suggesting that this axis is driven by forest expansion. Negative loadings correspond to cereals, grasses and anthropogenic indicator herbs (e.g. *Amaranthaceae*, *Cichorioideae*). PC2 loadings show positive values for *Triticum* L. and *Poaceae*, and negative values for *Ambrosia* L. and *Cichorioideae* (ESM Fig. 4), referring to the altering intensity of anthropogenic land use.

In Pp-1 (46.5–54 cm, 1983–1992 ($\pm$ 6 years)), grass pollen (*Poaceae*) dominates (average 23.7%) with a peak at 48.5 cm (33.6%). Cereals are abundant (5.7%). Arboreal pollen is 30.5% on average. Charcoal accumulation rate averages 13,238 pieces cm⁻² yr⁻¹. Coccal green algae *Pediastrum boryanum* (Turpin) Meneghini reaches a maximum of 30.4%. Palynological richness is 33.2.

In Pp-2 zone (38.5–46.5 cm, 1992–1999 ($\pm$ 4 years)) *Poaceae* decreases to 17.1%. *Ambrosia* (16.2%) and *Amaranthaceae* (11.1%) increase. *Salix* and *Quercus* L. rise slightly to 6.8% and 12.3%, respectively, with total arboreal pollen at 37.7%. *Cannabis* L.-type pollen reaches 2.2%. *Pediastrum boryanum* increases to 52.6%. Microcharcoal accumulation rate remains stable. Palynological richness is 32.7.

In the Pp-3 zone (30.5–38.5 cm, 1999–2005 ($\pm$ 2 years)), arboreal pollen rises to 42.3% due to *Quercus* (14.1%). Herbs decrease to 54%. *Cannabis*-type pollen increases to 4.9%, *Amaranthaceae* decreases to 5.5%. *Pediastrum boryanum* reaches 75.5% at 36.5 cm. Palynological richness decreases to 31.3.

In the Pp-4 zone (18.5–30.5 cm, 2005–2013 ($\pm$ 2 years)), *Pinus* pollen increases to 17.2% by 2009, then decreases to 10.2% in 2012. *Poaceae*

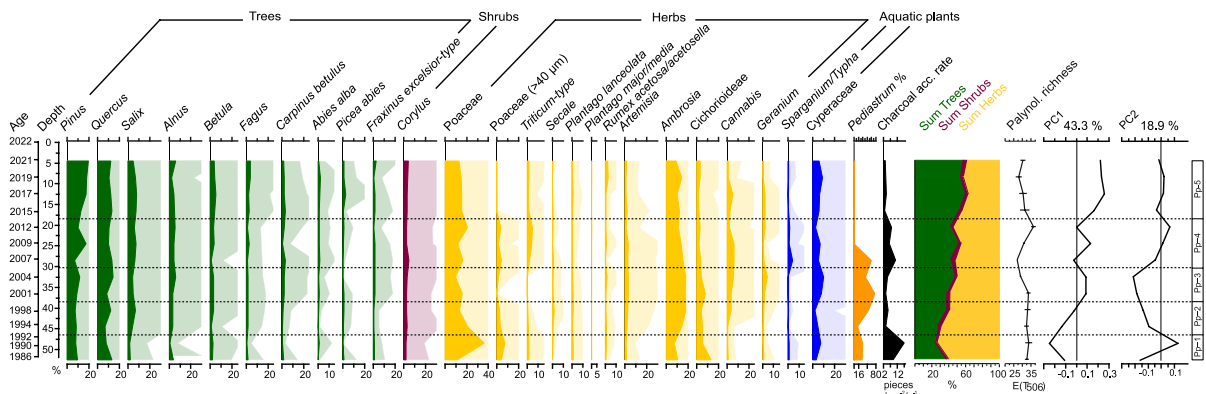


Fig. 5 Relative pollen frequency diagram (including the most common species with relative abundance > 2% and some selected pollen types) with the relative abundance of *Pediasium*, charcoal accumulation rates, palynological richness, PC1 and PC2 stratigraphic plot with the percentages of the total

variance the axes explain from the sediment of the Pátkai-t-2 gravity core from Pátka reservoir; for zonation optimal splitting by information content method was used; the pale colour indicates values increased by a factor of ten

pollen decreases to 11%, then rises to 19.7%. Total arboreal pollen reaches 45.3%. Cereals increase to 3.1%. Charcoal accumulation rate averages 8,185.8 pieces $\text{cm}^{-2} \text{yr}^{-1}$. *Pediasium boryanum* is present until 28.5 cm (2007) with 64.2% relative abundance, then disappears. Palynological richness reaches a minimum of 27 at 28.5 cm and increases thereafter.

The Pp-5 zone (0–18.5 cm, 2013–2022 (± 2 years)) was characterised by the rising arboreal pollen to 55.8% on average, mainly due to *Pinus* (16.4%), with slight increases in *Betula* (4.3%), *Salix* (6.5%), *Fraxinus excelsior* L. (1.6%), *Carpinus betulus* L. (2.24%) and *Fagus* (2.8%). *Poaceae* remains stable at 14.7%. Human-indicator weed *Rumex acetosa/acetosella* L. appears in the last two samples. Palynological richness is 30. Charcoal accumulation rate stabilises at 3,464.7 pieces $\text{cm}^{-2} \text{yr}^{-1}$.

Discussion

Integrated interpretation of environmental proxies and the drivers of chironomid community change in the reservoir

Using multi-proxy analysis from Pátka Reservoir, our intent was the interpretation of past environmental changes, comparing chironomid-inferred ecosystem responses, independent geochemical and palynological signals from the sediment, as well as water-quality

data collected by KDTVIZIG. The RDA analysis confirmed after stepwise selection that total sulphur (TS), geochemical PC1 (reflecting detrital input) and pollen PC2 (reflecting land-use intensity) are the most significant explanatory variables for the faunal changes (Fig. 6) and they all have a moderate, but significant influence on the chironomid community (the first two axes together explain about 45% of the variation in species composition). These relationships were further confirmed by the multi-decadal water-quality data (1986–2024, KDTVIZIG, Fig. 1b), showing a gradual eutrophication trend. This highlights the sensitivity of the chironomid assemblages to a combination of aquatic (bottom water oxygen regime) and terrestrial (catchment vegetation and land-use change) stressors and environmental pressure. While water temperature is typically the primary driver of faunal changes in pristine alpine and boreal lakes (Langdon et al., 2006; Nevalainen & Luoto, 2012), in shallow lowland lakes the main drivers more likely appear to be anthropogenic eutrophication, water management and sediment quality, as confirmed by our findings from Pátka Reservoir. The alignment between the variations of biotic and abiotic proxies (Fig. 6) supports the reliability of the chironomid analysis and supports the use of this macroinvertebrate group as an early-warning indicator of environmental degradation in artificial or natural aquatic systems.

The multi-proxy records from Pátka Reservoir reveal three distinct phases of environmental change

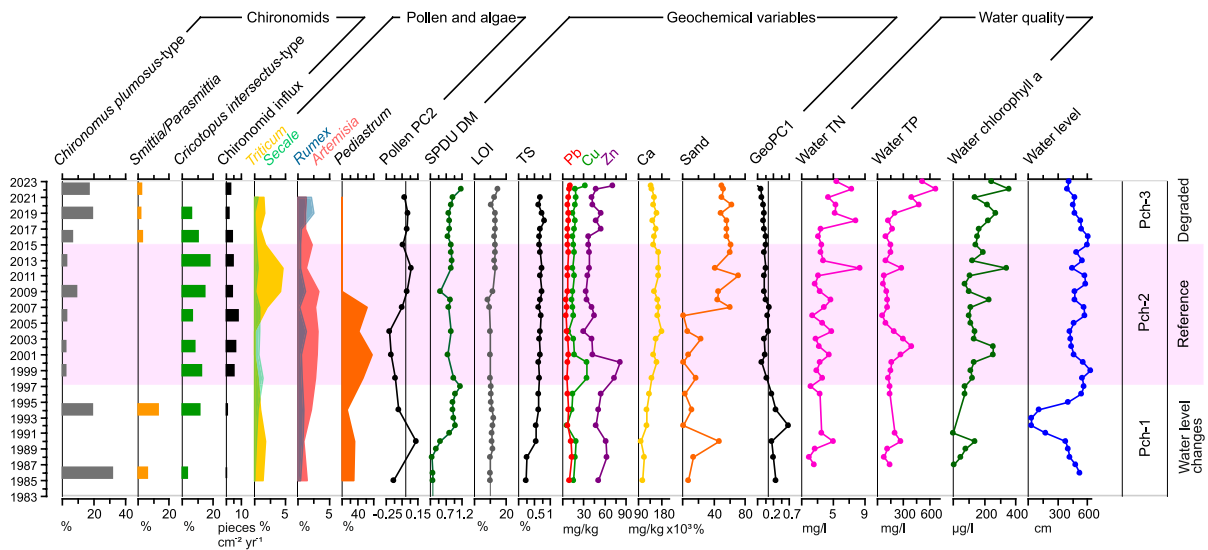


Fig. 6 Selected chironomid indicator species, cultivated and human indicator herbs, pollen PC2 stratigraphical diagram, geochemical variables, geochemical PC1 stratigraphic diagram, water-quality parameters, water level and chironomid

assemblage zones to support the temporal extent of the Pátka Reservoir reference state (pink shaded area in the figure covers inferred reference period)

over the last four decades. These shifts correlate closely with historical events (e.g. drainage and refilling of the reservoir in 1992 and 1994), water-quality variations and land-use changes.

Phase 1: Reservoir establishment and disturbance (1983–1997)

The basal part of the core (Zones Pg-1/2, Pch-1, Pp-1/2) corresponds to the period following the reservoir's construction (1974) and its complete drainage between 1991 and 1994. Geochemically, this phase is characterised by high concentrations of detrital elements (Fe, Al, K, Ti) and low organic content (LOI, TOC), indicating low productivity, intensive terrestrial soil erosion and clastic sediment input via the inlets due to the unstable hydrological regime (Cohen, 2003; Kylander et al., 2011). The high correlation between Mg and Al further supports the detrital origin of these elements (Norton et al., 1992). The geochemical record shows relatively high Fe and Mn concentration, referring to oxic sediment–water interface and low biological activity (Naeher et al., 2013).

The biological proxies reflect this instability and the hydrological shock of the drainage. The chironomid community is dominated by opportunistic

Chironomus species and semi-aquatic *Smittia/Parasmittia*, indicating fluctuating water levels and pioneer conditions. We assume that the dominant *Chironomus* taxon in this early phase was likely *Chironomus riparius* rather than *C. plumosus*, as the former is a rapid coloniser of new or disturbed habitats (Bitušik & Svitok, 2006), whereas the latter typically indicates established eutrophic standing waters (Langdon et al., 2006). The presence of rheophilic taxa (e.g. *Rheocricotopus effusus*-type) is likely a remnant of the lotic environment of the Császár Creek (Brooks et al., 2007; Andersen et al., 2013) that flowed through the basin during the drainage period. Normally, these rheophilous species are the first to colonise new habitats but disappear as they cannot survive lentic conditions (Nursall, 1952).

These observations align with the findings from other recently created or disturbed reservoirs in Central and Eastern Europe, where early colonisers dominated immediately after disturbance (Lucas et al., 2015b; Moreno and Callisto, 2006). Specifically, Csépes et al. (2007, 2012) linked the spread of *Chironomus* sp. to the establishment of the Kisköre Reservoir in Hungary.

Phase 2: Stabilisation and mesotrophic state (1997–2015)

Following the refilling (1994), the reservoir shifted towards a stable lentic environment (zones Pg-3, Pch-2, Pp-3/4), which represents the most favourable ecological phase in the record (Fig. 6). The decline in detrital elements and the coincident increase in Ca suggest a reduction in minerogenic input and the onset of biogenic carbonate precipitation, indicative of stable lake conditions.

This is strongly supported by the biological proxies. The stability promoted the expansion of submerged aquatic vegetation, as evidenced by the proliferation of macrophyte-associated chironomids (*Cladotanytarsus mancus*-type 2, *Polypedilum nubeculosum*-type, *Cricotopus intersectus*-type) (Brodersen et al., 2001; Gandouin and Franquet, 2002; Langdon et al., 2006; Andersen et al., 2013). The decline of pioneer taxa also supports the establishment of permanent lake conditions (Brooks et al., 2007). Indeed, water-level fluctuations slightly decreased during this period (Fig. 1b), which, together with reduced TP concentrations and stabilised dissolved oxygen levels, allowed the recolonisation of more sensitive taxa. Submerged macrophytes likely played a key role in maintaining clear water and well-oxygenated bottom conditions by stabilising sediments and providing habitat (Van Donk & Van de Bund, 2002; Thomaz & Cunha, 2010). This pattern resembles recovery processes observed in similar shallow reservoirs, such as Lake Müggel (Germany), where nutrient reduction measures led to ecosystem recovery (Hilt et al., 2018). The expansion of submerged vegetation has also been reported from the Solina Reservoir (Poland) and Lalla Takerkoust Reservoir (Morocco), where chironomid communities were enriched by macrophyte-associated taxa during periods of hydrological stability (Fleituch et al., 2002; Brooks et al., 2007).

The high taxon number and chironomid influx observed in this phase is likely linked to habitat heterogeneity; as noted by several authors (Ali et al., 2002; Papas, 2007; Tessier et al., 2008), there is a positive correlation between macroinvertebrate diversity and the complexity of aquatic vegetation. Morduchai-Boltovskoi (1961) suggested that the increase in population density is associated with the increased food resources that become available during the

decomposition of flooded terrestrial vegetation. The increasing proportions of *Secale* and *Triticum* suggest a shift towards anthropogenic land use, reflecting intensified agriculture in the catchment (Fig. 6). The peak in *Pediastrum boryanum* abundance further confirms a phytoplankton-dominated but balanced mesotrophic–eutrophic state with enhanced productivity but without severe pollution (Komárek and Jankovská, 2001; John et al., 2011). Although heavy metal concentrations (Cu, Zn, Pb) showed peaks around 1998–2000, these were likely associated with atmospheric deposition or urban wastewaters rather than natural sources (Itkonen et al., 1999; Meriläinen et al., 2003). Historical data suggest that during this period, despite moderate nutrient inputs, the ecosystem maintained a buffer capacity, likely supported by a balanced fish stock and moderate angling pressure (ESM Fig. 6). After 2009, the disappearance of green algae, accompanied by an increase in chlorophyll concentration (SPDU) and a higher sand proportion, suggests a less stable lake system characterised by enhanced eutrophication, increased energy input (likely due to changes in water management) and macrophyte proliferation. In the geochemical record (Fig. 3; ESM Fig. 1), a moderate increase in TS (total sulphur) indicates less oxygenated conditions at the sediment–water interface, likely due to enhanced organic matter deposition and decay.

Phase 3: Accelerated eutrophication and degradation (2015–2022)

The most recent period (zones Pg-4, Pch-3 and Pp-5) marks a distinct deterioration in water and sediment quality, driven primarily by intensified human intervention. The steep rise in TOC, TbN, TS and SPDU concentrations reflects intensifying organic enrichment and eutrophication. This organic overload led to benthic hypoxia, as evidenced by the increase in total sulphur (TS) and the collapse of the diverse, macrophyte-associated chironomid community. The fauna became depleted and shifted back to the dominance of *Chironomus plumosus*-type, a classic indicator of poor oxygen conditions, high trophic levels (Brodersen & Quinlan, 2006; Langdon et al., 2006) and ongoing ecological stress, replacing the sensitive vegetation-dwelling taxa. The dominance of *C. plumosus*-type under such conditions coincides with observations from other reservoirs, such as Banská

Štiavica (Slovakia) and the Szemes basin in Lake Balaton (Hungary), where its appearance indicated oxygen-depleted and nutrient-enriched conditions (Langdon et al., 2006; Magyari et al., 2024).

The disappearance of *Pediastrum* and the simultaneous increase in sedimentary chlorophyll products (SPDU) suggest a shift in the algal community structure, completing the transition to a hypertrophic state. Moreover, pollen signals also suggest that riparian vegetation expanded and aquatic vegetation with littoral habitats deteriorated. The pollen record still shows the rise of ruderal weeds which also confirms that the pressure on the reservoir ecosystem from land-use change has increased (Fig. 6). After 2015, the proportion of cereal pollen showed a decreasing trend near the pond (e.g. *Secale*, Poaceae > 40 mm). Changes in the Corine land cover database (Fig. 1b) indicate a 2.43% decrease in the proportion of agricultural land in the Császár Creek's watershed by 2018 compared to 1990, supporting the land-use decline detected by the pollen study. The charcoal accumulation rate also peaks in this section, which may reflect increased local biomass burning and/or land clearance in the catchment of the reservoir (Fig. 5).

Additionally, the repeated increase in lithogenic elements (Al, K, Ti) in the uppermost layers might be linked to changes in hydrology. Elevated heavy metal concentrations (e.g. Pb, Zn) (Fig. 6) suggest possible heavy metal contamination in the catchment, further stressing the ecosystem.

The primary driver of the eutrophication appears to be in-lake management rather than catchment agriculture. While land-use data indicate a slight decrease in agricultural area, records from the Federation of Fishing Associations (HOFESZ) reveal that the years of most conspicuous degradation in the reservoir's water quality were also remarkable regarding fish stocking and fish catches: approximately 20,000 kg of fish, mainly common carp (*Cyprinus carpio* L.), was stocked in 2016 and 2018, and recorded catches were close to this level in 2020. Interestingly, in the early 2000s, the initial period of nutrient loading of the reservoir after recharge, HOFESZ documented similarly high annual catches (around 20–25,000 kg), suggesting a connection between the high fish population, popular angling (and consequently large amounts of fish bait) and the increase in nutrients in the reservoir. During the period of approximately 2000–2015, although data gaps across several years

made interpretation challenging and uncertain, lake conditions were considered relatively stable. Annual fish catches peaked at around 6000–7000 kg, suggesting that a stable ecological state was associated with moderate levels of angling activity (ESM Fig. 6).

The recently established fish ponds along Császár Creek (ESM Text 1), including the Göbolyvölgyi Reservoir (2019, 5 ha), the Móric Major Irrigation Reservoir (2017, 44.8 ha) and their subsequent filling with water may have contributed to increased water retention and, consequently, to the eutrophication process. Additionally, treated sewage water entering the reservoir via Császár Creek also contributes to the elevated nutrient levels.

The chironomid assemblages of the last 40 years in Pátka Reservoir reflect a shift from the disturbed state of early colonisation after the drainage, through a transitional phase of mesotrophic stability towards recent degradation and increased trophic conditions. A similar transition is detectable in other temperate reservoirs that are exposed to anthropogenic and environmental pressures, including land-use change, nutrient loading and climate-variability. The trend observed from Pátka Reservoir also highlights the need for targeted water management interventions such as dredging, nutrient input reduction and improved catchment practices.

Restoration measures and strategic role of the Pátka Reservoir in replenishing Lake Velence

For water transfer from Pátka Reservoir to Velence Lake to be ecologically sustainable, the quality of the supplied water must not endanger the recipient ecosystem. Therefore, it is essential to identify a historical reference period in Pátka's sediment record that represents a target ecological state for restoration (Moss et al., 1996; Dearing et al., 2014).

Based on chironomid fauna, geochemical proxies and paleoecological data (Fig. 6), the period between 1997 and 2015 emerges as the most favourable ecological phase. During this time, mesotrophic-associated chironomid taxa were dominant, indicating stable, vegetated habitats and moderate trophic conditions (Brodersen & Quinlan, 2006; Eggermont & Heiri, 2012). Increased SPDU and TOC values suggested enhanced primary production without severe eutrophication, while pollen PCA results reflected a moderate level of anthropogenic influence in the

catchment. This period represents a balanced ecosystem state, with a stable benthic community structure and favourable oxygen conditions. In contrast, the post-2015 period shows significant ecological degradation, while the initial (1983–1997) phase was characterised by disturbed, unstable conditions. Thus, we designate the 1997–2015 interval as the reference state, serving as a realistic and ecologically sound, sustainable restoration target (Smol, 2008; Dearing et al., 2014). In addition to its water replenishment role, Pátka Reservoir is part of Hungary's National Ecological Network, functioning as an ecological corridor, which highlights the area's importance in supporting regional biodiversity. Restoring and preserving the ecological corridor role also aligns with the long-term sustainability of ecosystem services provided by the reservoir. In our opinion, improving the ecological condition of Pátka Reservoir is feasible through targeted catchment and in-lake management actions (Jeppesen et al., 2005), the means of which we summarised in Table 1, along with their applicability, cost and limitations.

Our multi-proxy results indicate that recent degradation trends, marked by declining chironomid diversity, increased sulphur concentrations and eutrophication signals, are primarily driven by fish-stocking-related nutrient loading leading to sediment oxygen depletion. This has been further amplified since 2017 by the construction of upstream fish ponds along the Császár Creek. Achieving good ecological status in accordance with the EU Water Framework Directive (WFD) (European Community, 2000) and reversing the observed negative trends requires a complex, multi-level restoration approach. This should include the improvement of sediment quality, the restoration of key ecosystem functions and the treatment of both point and diffuse pollution sources in the Pátka Reservoir catchment (European Community, 2000; Grizzetti et al., 2016) (Table 1). The mitigation of external nutrient sources should be implemented through the establishment of buffer zones, improvements in wastewater treatment, reducing untreated runoff from settlements and fish ponds in the Pátka Reservoir catchment (ESM Text 1) and the application of sustainable agricultural practices (cover crops and improved fertiliser management) (Sharpley et al., 2006; Kronvang et al., 2008). Buffer strips and riparian vegetation effectively reduce nutrient and sediment inputs by enhancing filtration and

promoting nutrient retention in soils (Mayer et al., 2007). In Pátka Reservoir, fish baits used by anglers and formulated feed for fish also stand as a significant external source of nutrients, which should be limited.

Given the poor ecological status of the Pátka Reservoir, the drawback of external nutrient loading alone might not be sufficient for the expected results. In these cases, internal nutrient loading must also be mitigated, for which sediment dredging represents a key intervention (Søndergaard et al., 1999) (Table 1). Dredging involves the removal of the nutrient-rich surface sediment layer, which reduces phosphorus release from sediment, increases the chances of macrophyte reintroduction and improves light transmission. However, studies from different regions confirm that dredging alone does not ensure long-term success but can be effective as part of an integrated restoration program (Søndergaard et al., 2007; Cooke et al., 2016). In Lake Vesijärvi (Finland), in addition to dredging, biomanipulation, reduction of external nutrient loading and management fishing led to significant improvements in water quality and ecosystem balance (Anttila et al., 2013; Salminen et al., 2021). Similarly, in Lake Durowskie (Poland), the sustainable restoration involved a combination of hypolimnetic aeration, phosphorus inactivation and biomanipulation, which effectively improved oxygen conditions and water transparency (Gołdyn et al., 2013; Dondajewska et al., 2019). Danish and Dutch lake restoration projects also confirmed that dredging had a positive short-term impact on water quality, but long-term success depended on concurrent external load reductions and other biomanipulation measures (Søndergaard et al., 2007; Bakker et al., 2013). Examining the specific impact and effectiveness of dredging, the case of Lake Trummen (Sweden) stands out: sediment removal successfully eliminated cyanobacterial blooms and restored ecosystem stability (Cronberg, 1982; Pettersson, 1998). Conversely, in China's Suyahu Reservoir, dredging alone failed to produce long-lasting improvements due to ongoing high external nutrient inputs (Gao et al., 2019). These examples highlight that the effectiveness of dredging often depends on other supplementary measures. The sediment of Pátka Reservoir was never dredged for financial reasons. In the reservoir, current data, including enhanced sediment nutrient concentrations and benthic hypoxia, support the necessity of periodic dredging (Fig. 6; Table 1) to reduce internal nutrient

Table 1 Summarised description of the recommended restoration measures for Pátka Reservoir

Method	Description	Advantages	Limitations	Cost	Applicability in Hungary	Ecological impact	Duration	Key references
Reduction of external nutrient sources	Reducing phosphorus and nitrogen from agriculture and settlements	Long-term and sustainable	Slow ecosystem response; requires cooperation	\$\$\$	High (supported by EU and national programs)	Stabilizes water quality, indirectly supports macrophytes	Long-term (> 10 years)	Jeppesen et al. (2005) and Søndergaard et al. (2007)
Internal phosphorus inactivation	Use of Al, Fe compounds or bentonite to bind P in sediment	Rapid P reduction; reduces algal blooms	Temporary; ecological toxicity is possible	\$\$\$\$	Medium/High (in shallow systems)	Lowers phytoplankton, indirectly supports macrophytes	Short to medium-term (3–10 years)	Cooke et al. (2016)
Macrophyte reintroducing	Reintroduction of native aquatic plants (planting or seeding)	Improves habitat; increases clarity	Success depends on light & nutrient; threatened by herbivores	\$\$	High (at local scale)	Provides structure, stabilizes sediments	Medium to long-term (5–10 years)	Hilt et al. (2006)
Biomaniipulation	Introduce predatory fish populations to control zooplanktivores	Nature-based solution; reduces algal blooms	Short-term; survival of predators not guaranteed	\$\$	Medium	Indirect effects via trophic cascade	Short-term (1–5 years)	Jeppesen et al. (1997) and Hansson et al. (1998)
Shoreline rehabilitation	Reconstructing reed belts, marsh zones and riparian vegetation	Supports biodiversity; natural filtration	Space-demanding; slow successional dynamics	\$\$	High (in suitable shoreline areas)	Improves habitats, filters nutrients, reduces erosion	Long-term (> 10 years)	Coops & Hosper (2002)
Dredging	Selective removal of P-rich or anoxic sediment layers	Immediate reduction of internal nutrients	Costly, ecological disturbance, sludge management necessary	\$\$\$\$	Medium (in hotspots)	Ecologically disturbing	Medium-term (5–10 years)	Steinman and Ogdahl (2008)
Hydromorpho-logical interventions	Modifying flow patterns, basin morphology or channel structure	Enhances water exchange and habitat complexity	Requires engineering; may conflict with water management	\$\$\$	Medium (with integrated planning)	Generally improves macrophyte and fish habitats	Long-term (> 10 years)	Blindow (1992) and Coops & Hosper (2002)
Monitoring, adaptive management	Long-term tracking of water quality, sediment and ecosystem	Enables targeted actions	For sustainability	\$	High (essential for any restoration)	Risk reduction, knowledge-driven planning	Continuous (supporting role)	European Community (2000)

Cost: \$ = low; \$\$ = medium; \$\$\$ = high; \$\$\$\$ = very high; Applicability in Hungary: high—legally feasible, technically available; medium—regulatory/technical limits

loading. However, for long-term success, dredging should be applied together with external load reductions and additional methods of ecosystem restoration (Istvánovics et al., 2007).

Biomanipulation measures (Table 1), such as reintroducing submerged macrophytes, reinforcing predatory fish and controlling benthivorous fish populations, may also help stabilise ecological functions and support chironomid community recovery (Blindow, 1992; Jeppesen et al., 1997; Hansson et al., 1998; Søndergaard et al., 2007; Hilt et al., 2018) in Pátka Reservoir. Submerged macrophytes play a key role in improving water clarity, reducing internal nutrient loading and providing habitat for diverse benthic invertebrates, including chironomids (Van Donk & Van de Bund, 2002; Thomaz & Cunha, 2010). Given the declining diversity and the abundance decrease of macrophyte-associated taxa in the uppermost layers, restoring littoral vegetation and limiting shoreline disturbances are particularly important for enhancing habitat complexity and facilitating the recovery of sensitive chironomid species (Table 1). Empirical studies confirm that successful littoral zone restoration can lead to increased benthic invertebrate diversity and improved ecosystem resilience (Bakker et al., 2013).

Fish community management further supports the improvement of the ecological status of a waterbody, macrophyte recolonisation and sediment stabilisation (Miller & Crowl, 2006) (Table 1); therefore, its implementation would be a priority for improving the water quality of the reservoir. Water bodies with an unbalanced fish stock and food chain often suffer from nutrient loading. This has a positive impact on the cyprinid fish population (which is the most numerous in the Pátka Reservoir), whereas for the predatory fish, these conditions are less favourable. The primarily cyprinid-dominated fish stock causes the excessive turning and mixing of bottom sediments (bioturbation) when the fish search for food. This promotes nutrient release, and thus has a negative effect on the oxygen regime of the water body. Cyprinids also tolerate conditions with less oxygen compared to predatory fish, leading to even more unbalanced fish stock and further deterioration of the ecological status of the waterbody. Management fishing, particularly the reduction of benthivorous species such as carp, can mitigate the internal nutrient loading through reduced bioturbation. As a result, the relative biomass

of certain predatory fish (e.g. perch, pike) increases. The beneficial impact can be seen on all levels of the ecosystem: decreasing benthivorous fish stock generally increases zooplankton biomass since fingerlings feed on zooplankton. According to the top-down effect, zooplankton feed on phytoplankton thus, the increased population of zooplankton reduces the growth of algae (phytoplankton), making the water less turbid and more clear (Hilt et al., 2006). Also, cyprinid fish themselves contain a relatively high amount of phosphorus (0.7–0.8% of fresh weight) (Mäkinen, 2008). Therefore, the stock reduction of these species also reduces the amount of phosphorus in the water body. At present, the drained state of the reservoir prevents the use of such measures, but after an eventual refill our proposal for KDTVIZIG is to replace the predominantly carp stock with a higher proportion of predatory fish stock. Unfortunately, this will not be an intervention that is easy to implement, as the lake, which has been used for recreational purposes since its creation, is very popular for angling and based on the available stocking data from HOFESZ, anglers prefer cyprinids.

Another issue for improving water quality and mitigating environmental pressure in the vicinity of the lake is that the level of protection is low around the Pátka Reservoir. The source and upper sections of Császár Creek are included in the Vértes Landscape Protection Area and the area north of the reservoir is part of the Natura 2000 sites (SPA, SCI) of the Zámolyi Basin, but the reservoir itself is not included in these protected areas (Kovács, 2014). The declaration of the Pátka Reservoir as a local protected area would significantly increase the opportunities for restoring the reference state. Thus, the future role of Pátka Reservoir in the water replenishment of Lake Velence must be assessed with caution. While the reservoir has the potential to provide supplementary water during low lake levels, the current ecological status does not meet the WFD “good” status (European Community, 2000). Transferring nutrient-rich water from Pátka to Lake Velence may increase eutrophication risks in the recipient system as elevated nutrients and suspended sediment loads can support algal blooms and oxygen depletion (Søndergaard et al., 1999). Therefore, ecological remediation and long-term monitoring should precede any large-scale hydrological interventions (Grizzetti et al., 2016) (Table 1). In summary, improving the

ecological condition of Pátka Reservoir is feasible through targeted catchment and in-lake management actions, as demonstrated by successful restoration efforts in similar systems (Søndergaard et al., 2007; Bakker et al., 2013). With adequate coordination between water authorities, angling federations and land users, the reservoir could move towards WFD-compliant status and contribute reliably to regional water security without threatening the ecological integrity of Lake Velence.

Conclusions

The multi-proxy analysis of Pátka Reservoir has revealed that the benthic chironomid community structure, in combination with geochemical and paleoecological indicators, provides a reflection of ecological changes over the past four decades. Three distinct faunal phases were identified, with alignment to historical events, catchment land-use changes and shifts in trophic status.

The *Chironomus plumosus*-type from the uppermost zone emerged as a robust indicator of degraded ecological conditions, including hypoxia and eutrophication, while the presence of mesotrophic and macrophyte-associated taxa (e.g. *Cladotanytarsus manicus*-type 2) characterised a more balanced ecosystem status. Pollen composition, geochemical variables and water-quality proxies closely correlated with benthic community structure, validating the applicability of the chironomids and the multi-proxy approach in reconstructing the ecological status of artificial water bodies.

Based on the multi-proxy evidence, the period between 1997 and 2015 was identified as the reference state for the suggested restoration of the Pátka Reservoir. During this interval, water quality and benthic community structure reflected stable, mesotrophic conditions, with moderate anthropogenic pressure and relatively healthy oxygen dynamics. This reference period represents an ecologically acceptable baseline for future restoration targets, particularly regarding the potential use of the reservoir for the water replenishment of Lake Velence. Achieving this reference state requires a holistic restoration

approach, combining reduction of external nutrient inputs, internal load mitigation through sediment dredging, habitat rehabilitation and biomanipulation. International case studies of water bodies with similar ecological conditions underline that dredging is effective only when integrated with catchment-scale management measures.

The current ecological degradation trends in Pátka Reservoir pose a significant risk to both the local ecosystem and the quality of water supplied to Lake Velence. Therefore, targeted, long-term interventions are essential to reverse these trends and achieve the “good ecological status” as defined by the EU Water Framework Directive. In summary, Pátka Reservoir holds the potential to serve as a valuable water resource for Lake Velence, but only if its ecological state is significantly improved. The restoration measures proposed in this study, guided by the defined reference state, provide a feasible pathway towards sustainable water management and ecosystem recovery.

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Data availability Please find the published data attached as an electronic supplementary data file (ESM Data Patka). The datasets generated during and/or analysed during the current study are also available from the corresponding author.

Declarations

Competing interests The authors have no competing interests to declare that are relevant to the content of this article.

Ethical approval No approval of research ethics committees was required to accomplish the goals of this study.

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