Silva Garbeta	vol. 2	p. 155–174	Vimperk, 1998

Limnological study of Plešné Lake in 1994–1995

Limnologická studie Plešného jezera v letech 1994–1995

Josef Hejzlar[1], Jiří Kopáček[1], Jaroslav Vrba[1], Renata Čížková[2], Jaroslava Komárková[1] & Karel Šimek[1]

[1] *Hydrobiological Institute, Academy of Sciences of the Czech Republic,*
Na sádkách 7, CZ–370 05 České Budějovice, Czech Republic
[2] *Department of Water Technology and Environmental Engineering,*
Prague Institute of Chemical Technology, Technická 5, CZ–166 28 Prague 6, Czech Rep.

Abstract

Chemistry and biology of water, and sediment composition were studied in an acidified (pH 4.5–5.0) mountain lake with forested catchment in south-western Bohemia. The comparison of water chemistry in tributaries and the lake showed that in-lake processes decreased acidity of inflowing water and its concentrations of dissolved organic matter (DOM), nitrate, total nitrogen, phosphorus and aluminium, and increased the concentration of ammonium. Seasonal stratification patterns corresponded to dimictic eutrophic lakes with anoxia in hypolimnion during both winter and summer stratification periods, however, the concentrations of chlorophyll *a*, phytoplankton and total phosphorus corresponded to meso- or oligotrophy. No release of dissolved reactive phosphorus from lake sediments was detected even during deeply anoxic conditions. Phosphorus fractionation in a surficial sediment sample demonstrated that sediment P was predominantly combined with hydroxyaluminium species and, thus, not liable to be released during changes of redox conditions. Ca, Al, Zn, Cu, and Cd concentrations showed vertical differences along the sediment vertical profile demonstrating acidification of the lake in the last decades. Acid-tolerant species of green algae (*Koliella*, *Carteria*, *Chlamydomonas*), dinoflagellates (*Peridinium*), cyanobacteria (*Pseudanabaena*), and several chrysophytes (only in spring) dominated in the phytoplankton community. High alkaline phosphatase activity and low ratio between chlorophyll *a* and phytoplankton biomass indicated severe limitation of phytoplankton by phosphorus. Macrozooplankton were sparse consisting of low densities of two crustaceans, *Heterocope saliens* and *Acanthocyclops vernalis*, and few rotifers. Bacterioplankton were likely responsible for partial decomposition and utilisation of polymeric DOM, serving also as a food source for protists.

Key words: forest lake, acidification, nutrients, phosphorus, metals, sediments, phytoplankton, bacteria, ectoenzymes

Introduction

The lakes in the Bohemian Forest (Šumava Mts.) represent unique natural aquatic ecosystems outstanding with rare species and specific biological communities, e.g. presence of endemic submersed macrophytes (GUTZEROVÁ 1996) or simplified pelagic food-web structures (VRBA & al. 1996). Although the lakes are situated in natural reserves and are protected from direct human disturbances they have been seriously affected by anthropogenic impacts via atmospheric transport of sulphur and nitrogen oxides causing acidification of water during the last decades (VESELÝ 1996; KOPÁČEK & al. 1998).

Plešné Lake had obtained less scientific attention in the past than the other lakes of the Bohemian Forest maybe due to its relatively poorer biocenosis and absence of fish. After

World War II the access to this lake was strongly restricted, making any systematic research difficult. In this study we wish to pay at least partly the debt to this interesting and valuable aquatic ecosystem. The aim of our article was to review historical and recent limnological research of Plešné Lake and document its stratification regime, water and sediment chemistry, and biological variables which we measured between April 1994 and February 1995.

Historical and recent limnological studies

The first documented limnological investigation of Plešné Lake was done in 1872 by Frič (1873) who determined dominant crustaceans species. Besides three littoral cladocerans, *Simocephalus* sp., *Acroperus* cf. *harpae*, and *Pleuroxus truncatus*, two different *Daphnia* spp. and four copepods, *Cyclops abyssorum*, *Heterocope saliens*, *Paracyclops fimbriatus*, and *Acanthodiaptomus denticornis* were present in the lake at that time. The latter diaptomid species had been never more found in the lake since the Frič´ s times. Planktonic daphnids have become apparently extinct in this century. As far as we know, these species have never been found by any visit of Plešné Lake during last almost forty years. Novák (1968) reported some *Cladocera* (7 ind. l^{-1}, only in 2-m depth) in August 1961, however, he did not refer species. Most likely, he caught chydorids. He also found some scarce copepods (not determined) and rotifers. Another crustacean species, *Acroperus elongatus*, a typical dwelling of acidified lakes, has been usually observed in the littoral of Plešné Lake during the last decade (E. Stuchlík & J. Fott – pers. comm.). According to Spilka (1993), recent abundances of the copepods were very low. In all but one case, he observed less than 1 ind. l^{-1} (volume weighted average of vertical profiles) of both *H. saliens* and *Acanthocyclops vernalis* during two vegetation periods (1991–1992). He also recorded a rotifer peak (*Synchaeta oblonga*, 55 ind. l^{-1}) in May 1992. At the present time, the top invertebrate predator in Plešné Lake seems to be *Glaenocorisa propinqua propinqua* (*Heteroptera*), collected in 1990–1993 by Papáček & Soldán (1995). The lake represents the only refuge of this species in Bohemia.

At the end of the 19th century, the lake was also investigated for submersed plants. A rare macrophyte, *Sparganium affine* was found in the lake by Velenovský in 1877 (Polák 1895), however, it has not been observed since that time. L. Čelakovský, Jr., discovered the presence of an endemic fern, *Isoëtes echinospora*, in the southern part of the lake in 1892 (Polák 1895); its population have survived at the same site in the lake until present (Tomšovic 1979, Gutzerová 1996).

There is only sparse information about the lake water composition in the early investigations, being restricted to several statements on colour and transparency. The yellow-brown coloration of water was described by Frič in 1872 (Frič 1873), Wagner in 1896 (Wagner 1897), and Švambera in 1903 and 1906 (Kuchař 1939) while Novák observed a blue-green colour of the lake in 1961 (Novák 1968). Transparency was measured by Wagner and Švambera using a Secchi disc; the maximum value of 3.3 m was observed in October 1903 (Kuchař 1939). The next measurements of transparency were done only in 1960s by Procházková & al. (unpubl. data) – 2.0 m (August 24, 1960), 1.7 m (June 28, 1961), 1.5 m (August 30, 1961), and by Novák (1968) – 2.4 m (August 11, 1961) and 3.2 m (April 4, 1962).

Fish were present in Plešné Lake in the first half of the 19th century, as Wenzig & Krejčí (1860) stated that brown trout (*Salmo fario*) lived in all the Bohemian Forest lakes except for Rachel Lake. However, some decades later, Frič (1873) and Wagner (1897) were unable to find any fish in Plešné Lake. Some factors causing lethality of fish were likely present in the water already in their times because an introduction of trout to Plešné Lake in 1894 and 1895 (Anonymus 1895) apparently failed. The possibility of trout introduction into the lake was unsuccessfully tested a half century later (1958) by Smíšek & Vostradovský (pers. comm. –

cited in Veselý 1994). They revealed the primary cause of fish mortality the acidity of water which had pH values in different depths in the range from 4.6 to 5.4.

The pH measurements that were occasionally done in the 1960s showed the onset of acidification. Procházková & al. (unpubl. data) measured the pH values in surface samples 5.0 in August 1960, 4.1 in June 1961 and 5.4 in August 1961. Novák (1968) made two vertical profile measurements of pH in August 1961 and April 1962 and determined values between 5.1 and 5.3. However, his colorimetric measurement was carried out with bromthymol blue, which has the lower limit of determination at pH 6, thus, his values cannot be considered correct. Acidic water (pH 4.4) was determined also in August 1976 (Krásný & al. 1986). Regular measurements of pH and water composition, that have been done since 1984 by Veselý (1996), confirmed the acidified status of Plešné Lake. Veselý (1987a,b) and Fott & al. (1987) suggested atmospheric deposition of sulphur and nitrogen into the catchment and acidification of catchment soils as the primary cause of acidification of Plešné Lake as well as the other lakes of the Bohemian Forest. An overview of available chemical analyses since the 1960s (Table 1) documents the culmination of sulphate, nitrate, and Ca concentrations in the lake water in 1980s as a result of the culmination of atmospheric emissions in Central Europe in this period (Kopáček & al. 1998).

Bottom sediments of Plešné Lake were investigated for the first time in 1987 by Veselý (1988). A piston corer specially developed to diminish compression of sediment layers during collecting was used to take a 115-cm long sediment profile. The sediment was highly organic with water content of about 98% in the upper 0–15 cm layer and of about 90% below the 20-cm depth. Metal analysis showed significantly higher concentrations of Zn, Fe and Cd in the uppermost 15-cm layer. Pb was increased in the depths of 0–60 cm two to four times compared to a background level below. Mg showed a similar pattern like Pb but with much lesser increase towards the surface. Ca dropped suddenly in the top 0–15 cm layer to less than a half in comparison with deeper layers. Be did not show pronounced changes along the vertical profile. Schmidt & al. (1991, 1993) sampled the sediments in 1990 in the central part of the lake at the depth of about 16 m. An in situ-freezing corer was employed to take about 35-cm long sediment profile for paleolimnological determinations of diatoms, siliceous cysts, chrysophycean scales, and carbonaceous particles and a Kajak corer for sampling of another 35-cm long profile for the water content, loss on ignition, and metal analysis. A change of sediment composition at the depth of about 15 cm was demonstrated by most parameters determined. Diatoms, siliceous cysts and chrysophycean scales indicated a decreased pH of water

Table 1. – Changes of chemical composition of the surface samples of Plešné Lake since 1960s. Units: mg l^{-1} except for NH_4^+, and TP that are in μg l^{-1}. Values given in italics were measured by imprecise methods and are believed to be biased. (TP = total phosphorus, TN = total nitrogen, COD = chemical oxygen demand by dichromate method).

Date	pH	Na	K	NH_4^+	Ca	Mg	Al	Fe	Mn	SO_4^{2-}	NO_3^-	Cl^-	TP	TN	COD	SiO_2
6/28/61[1]	4.1	—	—	—	*1.4*	*0.9*	—	—	—	4	1.2	—	11	0.6	3.4	—
8/30/61[1]	5.4	—	—	—	*1.4*	*0.9*	—	—	—	—	0.6	—	7	0.4	6	—
4/4/62[2]	*5.4*	*1.8*	—	*500*	*2*	*1*	—	*0.3*	—	*8.6*	*2.2*	*2.5*	—	—	—	4.8
8/16/76[3]	4.4	0.8	0.3	110	2.8	0.5	—	0.12	0.06	9.5	2.7	1.4	—	—	—	4.6
8/8/86[4]	4.7	1.0	0.5	—	1.5	0.3	0.9	0.10	0.06	9.9	2.5	0.9	—	—	—	5.1
1990–94[5]	4.7	0.9	0.4	61	1.2	0.3	0.7	—	—	7.3	1.6	0.6	9	0.7	7.7	—

[1] Procházková & al. (unpubl. data); [2] Novák (1968); [3] Krásný & al. (1984); [4] Veselý (1987); [5] Vrba & al. (1996), averages from 14 samples taken during 1990–1994

in the recent period. The carbonaceous particles originating from high-temperature combustion of fossil fuels of the industrial era (approximately the last one hundred years) were found in the depth from 0 to 12 cm. Metals showed three different patterns with the major change at 15 cm: (i) Ca decreased towards the surface, (ii) Al, Fe and Mg moderately increased but were variable in the upper 15-cm layer, and (iii) Zn, Pb, Cu, Cr, Ni and Cd were significantly increased towards the surface. The sedimentation rate was estimated from the ^{137}Cs profile 5.3 ± 0.2 mm per year, which positions the year of 1963 at the 15-cm depth, however, this statement contradicts to the data on carbonaceous particles, which show that this depth corresponds to about the first half of the 19th century.

Study site

Plešné Lake (German name: Plöckensteiner See; location: 48°47′ N, 13°52′ E; Fig. 1) is situated in the south-eastern part of the Bohemian Forest (Šumava Mts.) at the altitude of 1090 m a.s.l. on the north-eastern slope of the Plechý Mountain (1378 m a.s.l.). It is of a glacial origin and was shaped to the present form during the last glaciation about 10,000 years ago (CHÁBERA 1975). With the surface area of 7.5 ha, volume of 6.2×10^5 m^3 and maximum depth of 18 m (ŠVAMBERA 1939) it represents the third largest and deepest natural lake of the Bohemian Forest in both the Czech Republic and Germany. The area of the catchment including the lake surface comprises 0,67 km^2. The bedrock of the catchment is formed predominantly by granite (two-mica, coarse-grained granite to adamellite, the so-called Plöckenstein granite; VESELÝ 1994). The soil cover consists of shallow acidic brown soils or podzols (depth of 10 to 30 cm; pH 3.8 to 4.5; SCHMIDT & al. 1993). More than 70% of the catchment area is covered with approximately 150 years old coniferous mixed forest with predominance of spruce, *Picea excelsa* (SOFRON & ŠTĚPÁN 1971). The natural moraine of the lake was compacted, heightened and equipped with regulation sluices during the building of the Schwarzenberg canal in 1789–1791 to enable floating timber (ANONYMUS 1990). Water level has been stabilised by a monk-type (overflow) outlet at 1089.31 m a.s.l. since 1988, i.e. 0.69 m below the upper side of the memorial stone that was mounted at the water side into the lake moraine in 1868.

Sampling and analysis

Sampling

The lake and two of its tributaries (PL-II-a and PL-III; Fig. 1) were sampled from April–May till November 1994 in approximately monthly intervals and in February 1995. No samples from the tributaries were taken in July because the surface flow ceased during the long period of hot and dry weather, and in February 1995 when they were inaccessible due to a high snow cover. The lake was sampled with a Friedinger sampler at a buoy installed in the area of maximum depth (Fig. 1).

The sampled water from both the lake and the tributaries was immediately filtered through a 0.2-mm nylon mesh for chemical and/or enzymatic analyses. Unfiltered samples from the lake were preserved with Lugol solution for phytoplankton analysis and with 2 % formaldehyde (v/v final conc.) for counting of bacteria and protozoa. Dissolved forms of some analytes (P, organic carbon, organic nitrogen, chemical oxygen demand, Fe, Mn, absorbance) were determined after filtration through the glass-fibre filters MN–5 (nominal porosity of 0.4 μm; Macherey-Nagel, Germany). Samples for the determination of anions (Cl^-, SO_4^{2-}) were frozen and kept at –20 °C until analysis. Samples for the determination of metals (Na, K, Mg, Ca, Fe, Mn) were acidified with nitric acid (0.5 ml per 100 ml of sample) and stored at 5 °C before analysis.

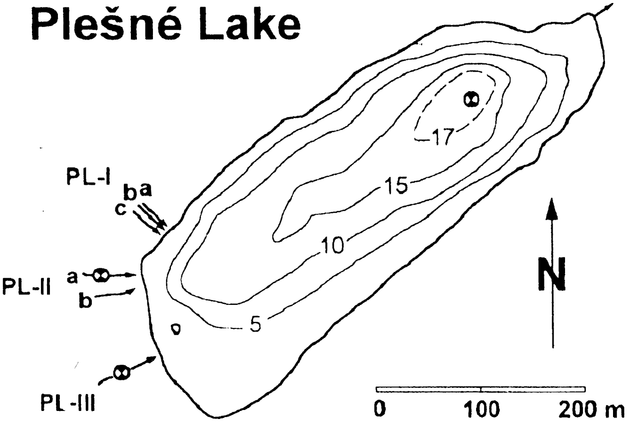

Fig. 1. – Bathymetry of Plešné Lake according to Švambera (1939) with the sampling stations at the lake and the tributaries in 1994–1995.

Two sediment cores were taken with a corer according to Hruška (1986) on August 26, 1994 near the buoy at the depth of 17 m. The cores were 5 cm in diameter and 30 and 15 cm long. They were transferred into the laboratory and there sectioned into 5.0 cm intervals. Sediment samples for the determination of total contents of organic carbon, nutrients and metals were lyophilised and passed through a 0.2-mm nylon mesh, not-sieved wet samples were used for the particle size distribution determination and phosphorus fractionation.

Chemical analysis of water samples

Temperature and dissolved oxygen (DO) stratification were measured by the OXI-meter (OXI 196, WTW, Germany). pH was measured with a combined glass electrode (GK 2321C, Radiometer). Acid neutralisation capacity to pH of approximately 4.5 (ANC) was measured by a Gran titration (Mackereth & al. 1989). The bis-pyrazolon method of Kopáček & Procházková (1993) was used for the determination of ammonia. Nitrite was analysed spectrophotometrically according to Bendschneider & Robinson (1952); nitrate was reduced to nitrite in a cadmium–copper column (Emteryd 1989, Collos & al. 1992) and determined by the same method. Total organic nitrogen (TON) and dissolved organic nitrogen (DON) were determined as a difference between Kjeldahl nitrogen (according to Procházková 1960) and ammonium concentrations. Sulphate and chloride were determined on the ion chromatograph AS 100 equipped with the pump P100 and the conductometric detector condoMonitor III (Thermo Separation Products, USA). Total sulphide was analysed with an iodometric titration method (APHA 1992). Dissolved reactive phosphorus (DRP) was analysed spectrophotometrically according to Murphy & Riley (1962); total phosphorus (TP) and total dissolved phosphorus (TDP) were determined with the same method after perchloric acid digestion (Kopáček & Hejzlar 1993). Chemical oxygen demand (COD) and dissolved chemical oxygen demand (DCOD) were determined by the dichromate semi-micro method (Hejzlar & Kopáček 1990). Total organic carbon (TOC) and dissolved organic carbon (DOC) were analysed by a wet-oxidation method with persulphate and UV-radiation in the LiquiTOC analyser (Foss/Heraeus, Germany). Absorbance (A_{254}) was measured at 254 nm with the Varian DMS300 spectrophotometer in a 1-cm cell and re-calculated to 1-m optical path. Total concentrations of Na, K, Mg, Ca, Fe and Mn were determined by optical emission spectroscopy (ICP) on the device PU 7450 (Leemans Labs Inc., USA).

Plankton analysis

Chlorophyll *a* concentration (Chl*a*) was determined after acetone extraction of samples fluorometrically (the whole-water technique, Vyhnálek 1994). Microscopic analysis of phytoplankton was done after sedimentation in Utermöhl chambers using an inverted microscope (Olympus IMT–2, Japan). To estimate the biomass and its composition, the computer program PHPL-HBI, developed in the Hydrobiological Institute AS CR České Budějovice was used. The biovolume of algae was expressed as a fresh mass (FM) under the assumption that 10^6 μm^3 of the biovolume is equivalent to 1 μg of FM.

Abundances of bacteria and heterotrophic nanoflagellates (HNF) were determined using epifluorescence microscopy (Olympus BH–2, Japan) after DAPI staining (Porter & Feig 1980). To evaluate a time-integrated grazing impact of all bacterivores (i.e. mostly HNF), 750-ml subsamples were incubated for 24 h at in situ temperature in the dark with an addition of fluorescently labelled bacteria (FLB; Sherr & Sherr 1993) constituting 3 to 6% of bacterial abundances. Subsamples (50 ml) for quantifying labelled (FLB) and natural bacterial cells were taken at times 0, 12, and 24 h. A linear model, taking into account also changes in natural bacterial abundances (Salat & Marrasé 1994), was used to calculate the rate of bacterial elimination. Bacterial production rate (P) was also estimated from the changes using the equation $P = (\ln N_T - \ln N_0)/T + E$, where N_0 and N_T were natural bacterial abundances at times 0 and T, T was the incubation period, and E was the bacterial elimination rate.

Ectoenzymatic activities were measured fluorometrically using 4-methylumbelliferyl (MUF) substrates (Hoppe 1993): 100 μmol l^{-1} (final conc.) of MUF-phosphate, MUF-α-glucoside, MUF-β-glucoside, and MUF-β-*N*-acetylglucosaminide (Sigma) for estimating phosphatase activity (PA), α-glucosidase activity (αGlcA), β-glucosidase activity (βGlcA), and β-*N*-acetylhexosaminidase activity (βGlcNAcA), respectively. Samples were incubated for 0.1–1 h (PA) or 9–12 h (other enzyme activities) at in situ temperature in the dark. Fluorescence was measured before and after incubation (Spekol 11 with fluorometrical device M4 FC 520, Zeiss, Germany) and corrected for respective blanks (autoclaved water).

Sediment analysis

Particle size distribution was determined by wet sieving of the sediment samples through the sieves of 1-mm and 74-μm mesh and by particle sedimentation rate analysis (Tuček 1983) of the fraction <74 μm. Contents of Na, K, Mg, Ca, Fe and Mn were determined by the same ICP method as for water samples but after digestion with nitric acid (EPA 1987). Total organic carbon was analysed by a combustion method in the LiquiTOC analyser (Foss/Heraeus, Germany), total nitrogen by the kjeldahlisation method (Procházková 1960) and total phosphorus by a spectrophotometric method after digestion the sediment samples with perchloric acid (Kopáček & Hejzlar 1995). Sediment phosphorus was fractionated according to Psenner & Puczko (1988) into the following fractions: (i) P dissolved in the interstitial water and loosely bound P (P_{H_2O}), (ii) reductant soluble P ($P_{Na_2S_2O_4}$), (iii) P adsorbed onto metal oxides ($P_{NaOH,20°C}$), (iv) P combined with carbonates and apatite P (P_{HCl}), and (v) organic P extractable with hot NaOH ($P_{NaOH,85°C}$).

Results and discussion

Water chemistry in lake and tributaries

The composition of water of the lake surface (Table 2) corresponded with the analyses from the last two decades (Veselý 1987, Vrba & al. 1996) and the downward trends due to the decreasing atmospheric deposition rates of sulphate and nitrate described by Veselý (1996) and Kopáček & al. (1998).

Table 2. – Transparency (Z_S), temperature (T), and chemical characteristics of the surface and near-bottom samples of Plešné Lake. (n.d. = not determined).

Parameter	Depth	4/29/94	5/16/94	6/15/94	7/20/94	8/26/94	9/19/94	11/3/94	11/29/94	2/16/95
Z_S, m	m	3.5	2.8	2.0	2.5	2.8	3.0	2.8	2.7	3.0
T, °C	0	3.7	12.6	12.3	19.6	14.9	11.4	5.9	3.7	0.8
	17	3.9	3.9	4.1	4.1	4.1	4.1	4.3	3.9	3.4
O_2, mg l^{-1}	0	10.9	10.5	10.1	7.8	8.5	9.2	9.3	10.9	11
	17	<0.1	3.7	<0.1	<0.1	<0.1	<0.1	<0.1	10.9	<0.1
COD, mg l^{-1}	0	11	12	8.6	8.9	5.6	7.6	7.0	7.0	8.2
	17	19	11	11	13	20	19	12	7.1	17
DCOD, mg l^{-1}	0	9.4	7.8	8.3	4.5	3	4	5.3	4.4	6.4
	17	12	6.3	8.4	8.2	12	13	8.2	5.2	11
DOC, mg l^{-1}	0	4.6	3.8	3.0	1.8	1.3	1.7	1.9	2.0	3.0
	17	6.1	3.1	3.3	3.8	5.5	5.0	3.9	2.4	5.4
A_{254} (1 m)	0	17	14	9.7	4.0	3.7	3.1	4.9	5.5	8.5
	17	20	10	10	11	14	19	11	5.5	23
pH	0	4.5	4.7	4.8	4.9	4.9	5.0	5.0	4.9	4.6
	17	6.0	5.0	5.3	5.54	6.0	5.9	5.7	5.0	5.8
ANC, mmol l^{-1}	0	–31	–14	–6	–10	–8	–6	–7	–4	–21
	17	17	3	12	25	128	111	50	–3	59
NH_4^+-N, µg l^{-1}	0	34	22	18	28	50	65	144	151	227
	17	650	385	495	552	1137	1110	640	153	574
NO_3^--N, µg l^{-1}	0	540	416	314	233	260	285	260	330	550
	17	26	288	50	18	10	14	15	350	71
NO_2^--N, µg l^{-1}	0	<1	<1	<1	<1	1	2	<1	<1	<1
	17	17	1	45	1	3	2	<1	<1	<1
TON, µg l^{-1}	0	315	360	317	320	245	273	250	257	250
	17	695	281	378	411	480	435	358	260	470
DON, µg l^{-1}	0	224	202	160	160	135	130	135	152	165
	17	n.d.	150	161	228	224	247	190	n.d.	290
DRP, µg l^{-1}	0	1.3	<1	<1	<1	<1	<1	<1	<1	<1
	17	<1	<1	<1	<1	<1	1.3	<1	<1	<1
TDP, µg l^{-1}	0	3.5	4.7	2.6	6.1	1.2	1.6	0.5	1.1	1.4
	17	3.9	2.2	3	2.9	4.1	4	2	2	1.6
TP, µg l^{-1}	0	9.7	13	8.3	12	6.8	8.4	5.6	6	6.8
	17	24	15	17	15	28	21	15	5.6	26
S^{2-}, mg l^{-1}	17	n.d.	n.d.	n.d.	n.d.	1.4	1.1	1.2	0.0	0.35
SO_4^{2-}-S, mg l^{-1}	0	2.3	2.3	2.2	2.3	2.4	2.3	2.4	2.5	2.1
	17	2.1	2.6	2.5	2.4	1.5	1.6	2.0	2.5	2.1
Cl^-, mg l^{-1}	0	0.59	0.55	n.d.	n.d.	n.d.	0.69	0.63	0.55	0.65
	17	0.68	0.57	0.57	0.62	0.62	0.65	0.55	0.54	0.65
Ca, mg l^{-1}	0	1.1	1.0	1.1	1.1	1.3	1.1	1.2	1.1	0.95
	17	1.2	1.1	1.3	1.1	1.4	1.1	1.3	1.1	1.2
Mg, mg l^{-1}	0	0.24	0.24	0.25	0.26	0.27	0.25	0.28	0.26	n.d.
	17	0.28	0.27	0.29	0.28	0.32	0.28	0.30	0.26	n.d.

Parameter	Depth	4/29/94	5/16/94	6/15/94	7/20/94	8/26/94	9/19/94	11/3/94	11/29/94	2/16/95
Na, mg l^{-1}	0	0.9	0.9	0.7	n.d.	0.9	0.8	0.9	0.6	1.0
	17	1.1	1.0	1.0	n.d.	1.0	0.9	0.9	0.6	1.1
K, mg l^{-1}	0	0.3	0.3	0.4	n.d.	0.4	0.3	0.6	n.d.	0.4
	17	0.3	0.3	0.7	n.d.	0.5	0.6	0.6	n.d.	0.4
Al, mg l^{-1}	0	0.83	0.82	0.76	0.61	0.73	0.58	0.60	0.55	0.51
	17	0.87	0.85	0.71	0.66	0.96	0.7	0.67	0.55	0.74
Fe, mg l^{-1}	0	<0.05	<0.05	0.05	0.05	0.19	0.07	0.08	0.06	0.05
	17	1.04	0.26	0.56	0.51	1.27	0.95	0.52	0.06	1.34
Mn, mg l^{-1}	0	0.04	0.04	0.04	0.04	0.05	0.04	0.05	0.04	n.d.
	17	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.04	0.05

The lake water significantly ($P > 0.95$) differed from the averages calculated from values of both tributaries (Table 3) in lower acidity, lower concentrations of dissolved organic matter (DOM), nitrate, total nitrogen, all phosphorus forms and aluminium. On the other hand, higher concentrations of ammonium were determined in the lake water than in the tributaries. These differences reflected various physical, chemical and biochemical processes occurring within the lake ecosystem (WETZEL 1975). Acidity was partly eliminated by denitrification and assimilation of nitrate and sulphate reduction above sediments. DOM decreased by photochemical degradation, microbial uptake, precipitation, and sedimentation processes. DRP was rapidly assimilated by phytoplankton and other microorganisms and, thus, transferred into the particles liable to sedimentation and resulting in the decrease of TP. Ammonium increased mainly due to the release from bottom sediments.

The composition of water in two sampled tributaries markedly differed, apparently due to the different character of their catchments. The tributary PL-II-a drained an area of a lesser slope with a developed spruce stand while PL-III drained a steep corrie wall with large boulders at its foot and only sparsely occurring trees and bushes. The temperature of PL-III was always lower indicating that the water passed through deeper underground profiles. Thus, the higher concentrations of DOM, phosphorus, Al and Fe and the lower concentrations of nitrate, Ca and Mg in PL-II-a in comparison with PL-III can be explained by a larger contact of water with soils rich in organic matter and an uptake by trees and soil microorganisms.

Stratification

The seasonal stratification pattern of temperature in Plešné Lake (Fig. 2) corresponded with that in other dimictic lakes of the temperate region (WETZEL 1975). The first sampling on 29 April fell into the end of spring mixing period. The temperature profile showed nearly complete destratification with 3.7 °C at the surface and 4.1°C at the bottom (Fig. 2A) but circulation did not occur within the total depth profile and the bottom layer remained stagnant and anoxic (Fig. 2B). Three weeks later the water column was already stratified. However, a complete homogenisation of the water body did not precede the formation of stratification as it was proven both by the dissolved oxygen concentration of 3.9 mg l^{-1} above the bottom (30% saturation) and the difference in water chemistry between the surface and bottom samples (Table 2). The stratification lasted until early November. The maximum temperature of epilimnion (19.6°C) was recorded during the July sampling. Bottom temperature only slightly increased from 4.0 to 4.3 °C during the whole stratification period. The thermocline was situated in the depths between 2.5 and 4.5 m until July and then, during the cooling of the surface layer, gradually deepened to 7.5 m in the middle of September and to 15 m in early

Table 3. – Physico-chemical characteristics of surface tributaries of Plešné Lake in 1994. (± CI = confidence interval of mean at 95% significance level; n.d. = not determined).

Parameter	Tributary	5/16/94	6/15/94	8/26/94	9/19/94	11/3/94	11/29/94	Average	± CI
Temperature, °C	PL-II-a	n.d.	5.5	9.7	8.5	5.0	n.d.	7.2	2.2
	PL-III	n.d.	2.0	6.0	5.5	4.1	n.d.	4.4	1.8
COD, mg/l	PL-II-a	15	16	18	18	26	15	18.0	3.4
	PL-III	8.0	8.3	12	8.2	8.0	5.5	8.3	1.6
DCOD, mg/l	PL-II-a	15	15	16	17	19	14	16.0	1.5
	PL-III	8.0	6.5	9.1	7.2	7.3	4.9	7.2	1.1
DOC, mg/l	PL-II-a	7.3	7.3	8.8	8.0	7.5	7.3	7.6	0.5
	PL-III	4.2	2.4	4.9	3.0	3.1	3.1	3.5	0.7
A_{254} (1m)	PL-II-a	31	29	33	30	27	27	29.5	1.9
	PL-III	16	7.4	17	9.5	10	9.6	11.6	3.0
pH	PL-II-a	4.3	4.1	4.1	4.2	4.2	4.2	4.18	0.07
	PL-III	4.5	4.7	4.4	4.6	4.6	4.6	4.57	0.06
ANC, mmol/l	PL-II-a	–58	–60	–97	–77	–85	–78	–76	12
	PL-III	–30	–17	–46	–31	–30	–22	–29	8
NH_4^+-N, µg/l	PL-II-a	10	15	0	18	24	11	13	7
	PL-III	23	26	26	68	10	0	26	19
NO_3^--N, µg/l	PL-II-a	517	354	1330	728	1120	1150	867	313
	PL-III	602	733	1570	937	1210	1275	1055	291
NO_2^--N, µg/l	PL-II-a	0.5	0.5	0.5	1	0.5	0.5	0.6	0.2
	PL-III	0.5	2	0.5	1	0.5	0.5	0.8	0.5
TON, µg/l	PL-II-a	n.d.	n.d.	n.d.	318	544	n.d.	431	221
	PL-III	n.d.	n.d.	n.d.	154	180	n.d.	167	25
SRP, µg/l	PL-II-a	26	25	24	26	22	19	24	2
	PL-III	15	7.0	15	9.4	9.6	8.0	11	3
TDP, µg/l	PL-II-a	n.d.	n.d.	n.d.	n.d.	24.8	23.4	24.1	1.4
	PL-III	n.d.	n.d.	n.d.	n.d.	n.d.	8.8	8.8	n.d.
TP, µg/l	PL-II-a	30	31	35	31	61	23	35	11
	PL-III	22	23	19	19	13	8.8	17	4
Ca, mg/l	PL-II-a	n.d.	n.d.	n.d.	n.d.	1.1	1.0	1.03	0.09
	PL-III	n.d.	n.d.	n.d.	n.d.	1.6	1.4	1.51	0.23
Mg, mg/l	PL-II-a	n.d.	n.d.	n.d.	n.d.	0.24	0.23	0.24	0.01
	PL-III	n.d.	n.d.	n.d.	n.d.	0.36	0.32	0.34	0.04
Al, mg/l	PL-II-a	n.d.	n.d.	n.d.	n.d.	1.3	1.1	1.21	0.15
	PL-III	n.d.	n.d.	n.d.	n.d.	1.0	0.9	0.95	0.08
Fe, mg/l	PL-II-a	n.d.	n.d.	n.d.	n.d.	0.19	0.03	0.11	0.16
	PL-III	n.d.	n.d.	n.d.	n.d.	0.05	0.05	0.05	0.01
Mn, mg/l	PL-II-a	n.d.	n.d.	n.d.	n.d.	0.05	0.05	0.051	0.001
	PL-III	n.d.	n.d.	n.d.	n.d.	0.06	0.05	0.054	0.008

November. The autumnal turnover of the water column with the equal chemistry of the surface and bottom samples was observed on November 29.

The concentration of dissolved oxygen (DO; Fig. 2B) fluctuated in the surface layer near the saturation value during the whole studied period. Slightly oversaturated values were obtained in the surface layer in the spring. The highest DO concentrations were measured in the metalimnion at the depths between 4 and 6 m until August with a maximum value of 13 mg l^{-1} (130% oversaturation) in July, apparently caused by the primary production of phytoplankton forming a similar metalimnetic maximum (Fig. 1C). In the hypolimnion, the low spring DO concentration started to decrease from the very beginning of summer stratification at a rate of ~ 0.10 mg l^{-1} d^{-1} and no measurable oxygen was present in the bottom layer already in June. During the autumn overturn the bottom concentration was replenished to 10.9 mg l^{-1} (94% saturation). The rate of oxygen depletion above the bottom during winter stratification was similar as in the summer period (~ 0.13 mg l^{-1} d^{-1}).

The concentrations of the other constituents which took part in the biochemical processes, i.e. compounds serving as nutrients, energy source and/or electron acceptors during oxidation of organic matter (NO_3^--N, SO_4^{2-}, NH_4^+-N, organic compounds) changed according to the patterns common in eutrophic lakes (WETZEL 1975, BUFFLE & STUMM 1994). Three distinct zones gradually developed within the depth profile after establishing stratification (Fig. 2, Table 2): (i) The epilimnion and metalimnion zone, where NH_4^+-N and NO_3^--N concentrations decreased as a result of phytoplankton assimilation, ANC and pH increased moderately mainly due to the bioconsumption of nitrate, DOC, COD and A_{254} decreased apparently by photo- and microbial decomposition of dissolved organic matter, and TP concentration decreased due to sedimentation of seston particles. (ii) The upper part of hypolimnion with relatively stable concentrations of all water constituents. (iii) The lower hypolimnion zone where the development of anoxia was followed by the consecutive depletion of NO_3^--N and a drop of SO_4^{2-} concentrations by microbial respiration, together with an increase of NH_4^+-N due to the release from mineralised sediment organic matter, S^{2-} from reduction of SO_4^{2-}, and ANC due to the consumption of H^+ ions in these processes. The concentrations of total and dissolved Fe also increased in this zone. TP concentration increased due to the gradual accumulation of colloidal and resuspended matter in the bottom layer during the stratification. Interestingly, DRP remained below the detection limit of the method (i.e. 1 μg l^{-1}) for the whole period.

Phytoplankton

The phytoplankton community consisted of several most abundant species of green algae and dinoflagellates and one species of cyanobacteria during all the season. No evident seasonal alternation of dominant species occurred which would be comparable with mesotrophic non-acidified lakes. The only difference between the spring and summer aspects was the spring occurrence of chrysophytes (*Chrysochromulina parva*, *Ochromonas* sp., *Chrysolykos complanatus*, *Chrysococcus rufescens*), which disappeared from the dominance in June. Green algae were represented by two or three species of *Koliella*, primitive filamentous algae, which persisted in the phytoplankton during all the season. The cells divided into two daughter cells, which remained together for some time forming short two-cell filaments. The sizes of cells were very different (lengths varied between 4 and 35 μm). The numbers of *Koliella* increased during the season as well as the share of green algae in the whole phytoplankton (another important portion were green flagellates *Chlamydomonas* and *Carteria*). Since May, dinoflagellates *Peridinium ueberrimum* and *Peridinium inconspiquum* made about 0.5–0.6 mg l^{-1} of FM which represented 6 to 20% of FM. A permanent part of the phytoplankton was formed by *Pseudanabaena limnetica* (forma) which was found in all the studied samples.

Both the distinct summer stratification and the presence of specific phytoplankters enabled

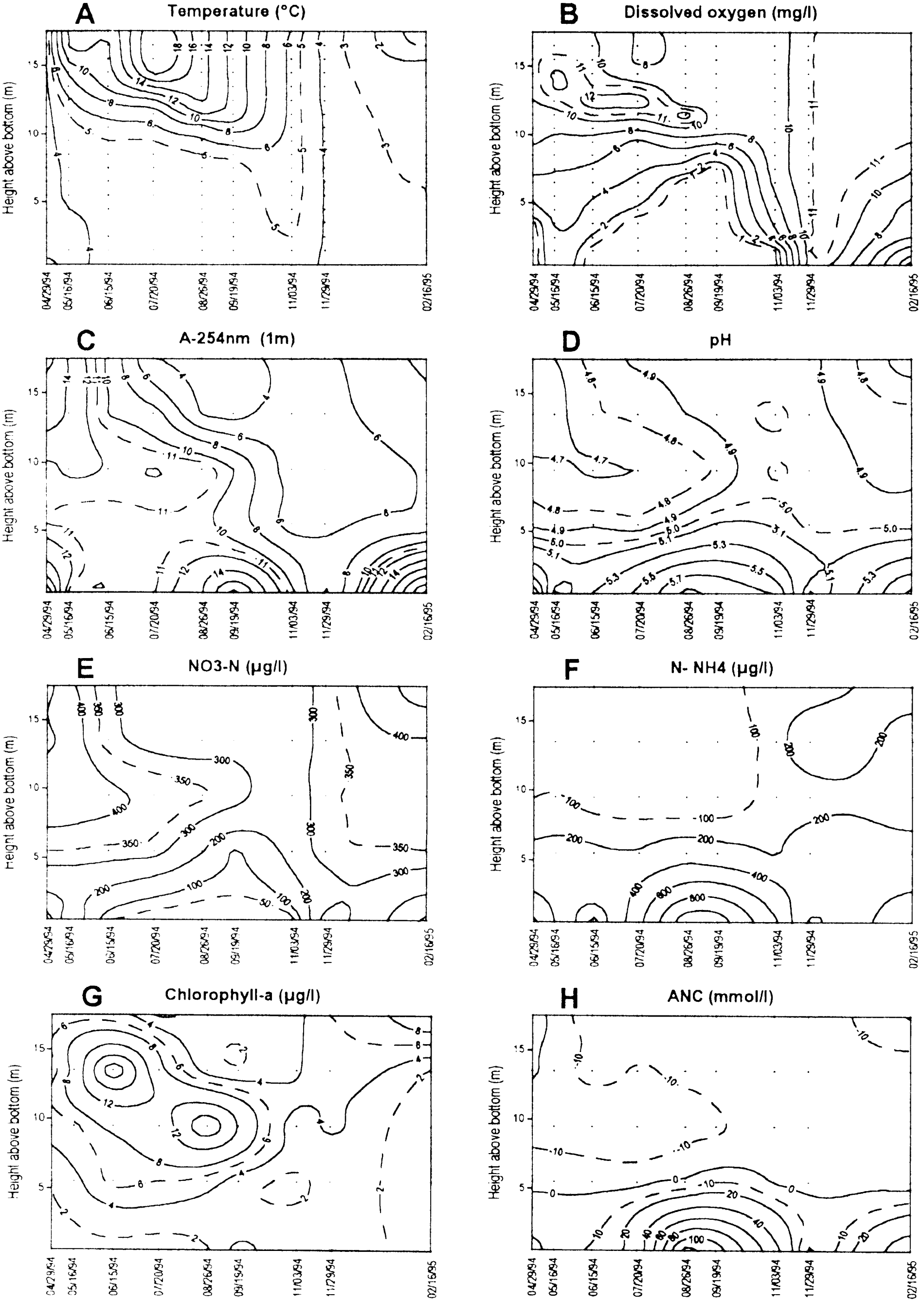

Fig. 2. – Vertical stratification of selected physical and chemical parameters in Plešné Lake during the whole study period: temperature (A), dissolved oxygen (B), absorbance at 254 nm (C), pH (D), concentration of nitrate (E), ammonia (F), chlorophyll *a* (G), and alkalinity (H).

conditions for the development of algal maxima in deeper layers (Fig. 2G). For example, on 20 July, phytoflagellates of the genera *Carteria*, *Peridinium*, and *Chlamydomonas* were the dominants in the 4-m depth, while *Koliella* dominated in the surface layer. In August, however, no significant difference was found between the samples from both layers. Recent preliminary measurements of primary production (J. NEDOMA – unpubl. data) suggest that metalimnetic maxima of phytoplankton in acidified lakes could be simultaneously controlled by two factors operating in parallel: dropping light intensity along with increasing concentration of inorganic C downward to the bottom (cf. also Fig. 2H). Such a light/CO_2 control might partly explain also the observed pattern of vertical distribution of the phytoplankton in Plešné Lake. On the other hand, extremely high PA (see below) also indicated severe phosphorus limitation of the lake phytoplankton.

The biomass of algae and cyanobacteria in the surface layer is given in Table 4 and vertical stratification of Chl*a* is plotted in Fig. 2G. No clear agreement in the courses of Chl*a* and FM was achieved. The seasonal maximum of FM in the surface layer was found on June 15 (7.8 mg l^{-1}), corresponding also to the seasonal minima of transparency and NH_4^+-N (cf. Tables 2 and 3). The phytoplankton biomass was also estimated in the 4-m depth in the last two summer samples, where FM was always higher than in the surface layer. The absolute maximum of FM was as high as 8.1 mg l^{-1} at 4 m on July 20. Also maximum Chl*a* was usually recorded at the 4-m (up to 23.6 μg l^{-1}) or 8-m depths during the summer stratification, although a winter Chl*a* maximum of 10.4 μg l^{-1} was observed just below the ice cover (Fig. 2G). A similar vertical distribution of autotrophic nanoplankton observed also NOVÁK (1968) in August 1961, with the maximum of 19.6 × 10^3 cell ml^{-1} at the 6-m depth. Corresponding algal abundances (without cyanobacteria) were slightly higher (range: 9–62 × 10^3 cell ml^{-1}) during summer 1994.

Table 4. – Phytoplankton biomass (FM), concentration of chlorophyll *a* (Chl*a*), abundance of bacteria and heterotrophic flagellates (HNF), and activity of phosphatase (PA), α-glucosidase (αGlcA), β-glucosidase (βGlcA), and β-hexosaminidase (βGlcNAcA) in the surface and bottom samples of Plešné Lake. (n.d. = not determined).

Parameter	Depth	4/29/94	5/16/94	6/15/94	7/20/94	8/26/94	9/19/94	11/3/94	11/29/94	2/16/95
FM, mg l^{-1}	0 m	1.0	4.8	7.4	2.7	2.4	n.d.	n.d.	n.d.	n.d.
Chl*a*, μg l^{-1}	0 m	5.1	5.1	4.7	1.5	2.5	2.5	3.4	6.0	10.4
	17 m	n.d.	1.5	1.6	1.1	1.7	4.8	2.6	4.1	0.5
Bacteria, 10^9 cell l^{-1}	0 m	n.d.	1.76	1.79	2.45	2.98	2.25	0.94	0.80	1.21
	17 m	n.d.	1.36	2.34	2.37	6.55	3.02	1.72	0.77	0.59/6.9[1]
HNF, 10^3 cell l^{-1}	0 m	n.d.	n.d.	447	87	291	373	435	353	67
	17 m	n.d.	n.d.	194	129	171	90	215	19	97
PA, μmol l^{-1} h^{-1}	0 m	n.d.	n.d.	6.7	5.5	4.0	5.5	0.70	n.d.	n.d.
	17 m	n.d.	n.d.	7.3	3.7	4.6	4.4	0.29	n.d.	n.d.
αGlcA, nmol l^{-1} h^{-1}	0 m	n.d.	n.d.	1.5	5.2	1.5	1.2	1.0	n.d.	n.d.
	17 m	n.d.	n.d.	5.4	7.8	2.1	4.6	2.5	n.d.	n.d.
βGlcA, nmol l^{-1} h^{-1}	0 m	n.d.	n.d.	7.0	15.6	5.3	7.2	4.6	n.d.	n.d.
	17 m	n.d.	n.d.	7.9	14.9	5.9	6.8	6.3	n.d.	n.d.
βGlcNAcA, nmol l^{-1} h^{-1}	0 m	n.d.	n.d.	1.8	10.6	2.1	1.9	3.3	n.d.	n.d.
	17 m	n.d.	n.d.	11.6	33.8	16.5	12.6	10.1	n.d.	n.d.

[1] a sample taken very close above the bottom

From the physiological point of view, the biomass of phytoplankton was found high in comparison with Chl*a*. The Chl*a* concentration in the biomass varied between 0.49 % in spring and 0.06% in summer. The percentage of Chl*a* in FM of healthy phytoplankton usually varies round 0.7–1.0% (DESORTOVÁ 1981). Such a low Chl*a* in the biomass can support a hypothesis that phytoplankton could not grow well in Plešné Lake, e.g. due to phosphorus and/or carbon limitation. This concerned mainly the species of the genera *Koliella* and *Peridinium*, which were dominant in summer. In spring, on the other hand, also green flagellates (*Chlamydophyceae*) were frequent, which commonly contain higher concentration of Chl*a* in biomass.

The lack of consumers prevented the development of a clear water phase typical for mesotrophic temperate lakes (SOMMER & al. 1986). Further effect, that can be expected, is a low turnover rate of phytoplankton and the presence of a special assemblage, which would never develop in the presence of grazers.

According to the gained data, it was proved that the phytoplankton biomass in Plešné Lake contributed seriously to the bulk of the organic matter. From the comparison of biovolume and Chl*a* estimations, more serious measure of the phytoplankton seems to be the biovolume (FM) than Chl*a*.

Bacterioplankton

Bacterial abundances in Plešné Lake varied within one order of magnitude (mean: 1.87×10^9 bact. l^{-1}, range: $0.59–6.90 \times 10^9$ bact. l^{-1}) during the study period. Local minima were usually recorded at 8 m during the summer stratification, except for August, when bacteria peaked in all layers (Fig. 3A). Low bacterial numbers were found in November and February (Table 4). On the other hand, maximum bacterial abundances were recorded in an anoxic layer above the bottom both in summer and winter (Fig. 3A, Table 4), where larger, morphologically distinct bacteria were usually observed, as well. Moreover, the most recent data on preliminary phylogenetic composition of the bacterioplankton (P. KOJECKÁ – unpubl. data on in situ hybridisation of fluorescently labelled r-RNA probes) have suggested that bacteria of different subclasses of Eubacteria are likely to be present in the epilimnion and the anoxic hypolimnion of Plešné Lake. Bacterial filaments were also present in some samples.

Ectoenzyme activities

The summer vertical distribution of PA (Fig. 3C) followed a similar pattern to that found for Chl*a* (cf. Fig. 2G) suggesting an algal origin of most of PA in the production zone of the lake. Specific PA, calculated per unit of Chl*a*, averaged 1.70 (± 0.95) μmol μg^{-1} h^{-1} for 0- to 8-m depths during the vegetation period (June – September), whereas dropped to 0.22 μmol μg^{-1} h^{-1} in November. Nevertheless, PA did not differ remarkably in either surface and bottom layer (Table 4). MÜNSTER & al. (1992) always observed algal PA below 1 μmol l^{-1} h^{-1} in the polyhumic, phosphorus limited Lake Mekkojärvi. In Plešné Lake, however, PA was as high as 22 μmol l^{-1} h^{-1} in the metalimnion. We suppose that the phytoplankton could balance a blockage of organic phosphorus by aluminium (JANSSON & al. 1986) in the acidified Plešné Lake by so extremely high PA.

General patterns of vertical profiles of αGlcA, βGlcA, and βGlcNAcA were very similar (cf. Figs. 3D–F) suggesting a concomitant bacterial decomposition of polymeric organic matter (e.g. starch, cellulose, chitin or peptidoglycan) during the season. All three ectoenzyme activities peaked in July (Table 4, Figs. 3D–F) despite of the maximum bacterial number in August. Another conspicuous phenomenon was a usually very high ectoenzyme activity in the anoxic layer above the sediment. Nevertheless, maximum polysaccharide hydrolysis appeared a month before the most anoxic conditions and the highest bacterial numbers (cf. Figs. 2B,E,H

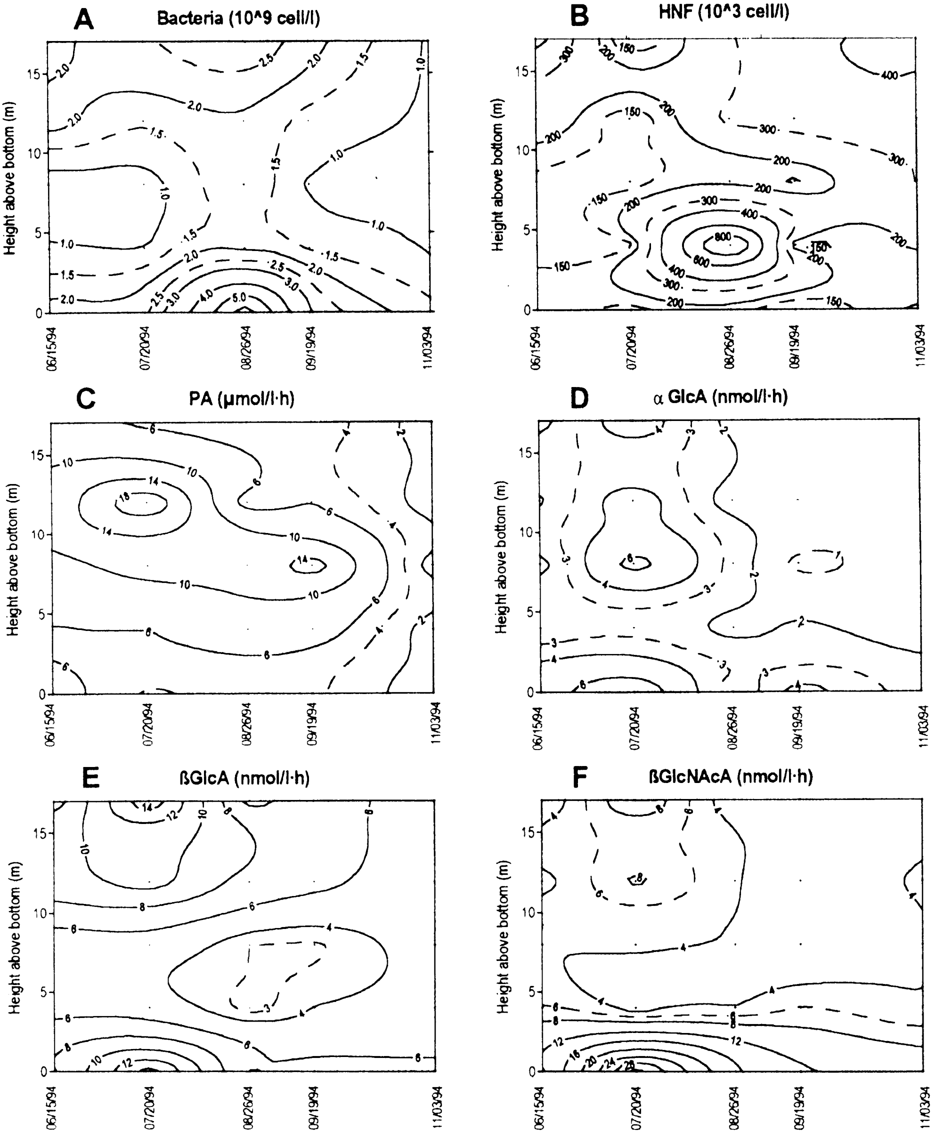

Fig. 3. – Vertical stratification of some microbial parameters in Plešné Lake during the vegetation period in 1994: abundance of bacteria (A) and heterotrophic nanoflagellates (B), activity of phosphatase (C), α-glucosidase (D), β-glucosidase (E), and β-hexosaminidase (F).

and 3D–F). Values of αGlcA and βGlcA in Plešné Lake were comparable with polyhumic and eutrophic waters (Münster 1991, Vrba 1992). However, the maximum βGlcNAcA values in the anoxic layer (Table 4) were usually even higher than in a eutrophic reservoir (Vrba 1992). There is no clear explanation of such a high βGlcNAcA above the bottom yet.

The low ratio of αGlcA:βGlcA (mean 0.42) apparently indicated prevailing bacterial degradation of recalcitrant plant material (e.g. cellulose-like) from the watershed (VRBA 1992) of Plešné Lake. MÜNSTER (1991) refers the average ratio of 0.39 for the polyhumic Lake Mekkojärvi. Besides phytoplankton primary production, allochthonous polymeric organic matter might be another important substrate source fuelling the microbial loop in Plešné Lake (see also TRANVIK 1992).

Protozoa

Abundances of HNF averaged 2.34×10^5 cell l^{-1} with relatively low temporal and spatial variability in Plešné Lake. Just two extreme values were found during the study period: a peak of 1.08×10^6 cell l^{-1} at 12 m on August 26 (Fig. 3B) and a minimum above the bottom on 29 November (Table 4). Ciliates, being usually very rare in Plešné Lake, were not quantified in the study.

Table 5 documents a potential grazing impact of bacteriovorous HNF on bacterioplankton in Plešné Lake. Between 6 to 41% of the bacterial standing stock was daily consumed according to the time-integrated estimate of the rate of bacterial elimination. Both the highest percentage of bacterial elimination and maximum bacterial "turnover" rate coincided with the HNF maximum. Surprisingly high elimination rate of bacteria was found even under anoxic conditions (Table 5).

Zooplankton and zoobenthos

Zooplankton were not sampled regularly during this study. Only two crustacean species, *Heterocope saliens* and *Acanthocyclops vernalis* being resistant to low pH and high aluminium content, and few rotifers (*Microcodon clavus*, *Brachionus urceolaris serricus*, *Keratella serrulata*) have been recorded in some samples in concordance with the catches in the last decade.

Among benthic littoral organisms (an autumnal sampling on November 3, 1994) prevailed *Nemurella pictetii* (*Plecoptera*), *Leptophlebia vespertina* (*Ephemeroptera*), and scarce caddisworms (*Oligotrichia striata*, *Plectrocnemia conspersa – Trichoptera*) (E. STUCHLÍK – unpubl. data). All but *O. striata* were observed in the early 1990s also by SOLDÁN & PAPÁČEK (pers. comm.). A mass occurrence of *L. vespertina* was reported from Plešné Lake by LANDA & SOLDÁN (1989).

Bottom sediments

Both sampled sediment cores appeared to be undisturbed with a clear sediment–water interface. They showed a distinct lamination with subtle layers 0.5 to 1 mm thick to the depth of approximately 7 cm and homogeneous deeper layers. The uppermost 10 cm had a brown colour, which became darker with increasing depth. The results of analysis of three selected

Table 5. – Vertical profiles of heterotrophic nanoflagellates (HNF) abundance, bacterial abundance and production, and elimination rate of bacteria by HNF grazing in Plešné Lake on August 29, 1994.

Depth, m	HNF, 10^3 cell l^{-1}	Bacteria, 10^9 cell l^{-1}	Bacterial production 10^9 cell d^{-1}	Elimination rate of bacteria by HNF, 10^9 cell l^{-1} d^{-1}	% per day
0	291	2.98	0.07	0.18	5.9
4	324	1.92	0.47	0.15	7.7
8	186	2.01	0.10	0.41	20
12	1080	1.80	0.62	0.74	41
17	171	6.55	1.70	1.17	18

layers of these sediment cores, 0–5 cm, 15–20 cm and 25–30 cm, are in Table 6. The sediment was highly organic with LOI of 75 % in the top layer and 65–68 % in the deeper layers. Total carbon corresponded to LOI. Particle size distribution did not change with the depth and the sediment could be classified as clayey silt with 65–72 % silt and 26–31 % clay.

Several parameters showed vertical differences, which evidenced that water chemistry of Plešné Lake changed in the past. Unfortunately, we are not able to conclude with certainty about the age of the analysed sediment layers. If the occurrence of carbonaceous particles in the uppermost 12-cm layer in the core of SCHMIDT & al. (1993) is considered to be a representative of the industrial era we may estimate the sedimentation rate of about 1 mm per year. With this very crude estimate, the top 0–5-cm layer originated from the last approximately fifty years. The apparent sedimentation rate in deeper layers is probably even lesser due to compaction. According to VESELÝ (pers. comm.), the 15–20-cm layer could originate from the 16th to18th centuries and the 25–30-cm layer was even approximately three hundred years older.

The difference in Kjeldahl N, which was by about 50% higher in the top layer, can be ex-

Table 6. – Characteristics of bottom sediments in Plešné Lake sampled on August 28, 1994. (n.d. = not determined).

Parameter	Sediment layer		
	0-5 cm*	15-20 cm	25-30 cm
Water content, %	97.8 ± 0.1	94.4	94.0
Loss on ignition, % DM	74.5 ± 0.5	65	68
Particle size distribution:			
> 1 mm, %	0.2 ± 0.1	0.3	0.1
74 µm - 1 mm, %	2.1 ± 0.2	4.0	4.3
2 µm – 74 µm, %	72 ± 5	67	65
< 2 µm, %	26 ± 5	29	31
Total carbon, % DM	34.5 ± 1.6	32.7	33.4
Kjeldahl nitrogen, mg g DM^{-1}	38.8 ± 0.2	24.4	24.6
Total phosphorus, mg g DM^{-1}	3.40 ± 0.05	3.6	6.1
Fractionation of TP:			
P_{H_2O}, % TP	18 ± 1	n.d.	n.d.
$P_{Na_2S_2O_4}$, % TP	6.9 ± 0.7	n.d.	n.d.
$P_{NaOH,20°C}$, % TP	73 ± 2	n.d.	n.d.
P_{HCl}, % TP	0.7 ± 0.1	n.d.	n.d.
$P_{NaOH,85°C}$, % TP	1.9 ± 0.4	n.d.	n.d.
Ca, mg g DM^{-1}	2.8 ± 0.3	4.8	6.2
Mg, mg g DM^{-1}	0.75 ± 0.10	0.9	0.8
Fe, mg g DM^{-1}	7.3 ± 0.3	12	9.7
Al, mg g DM^{-1}	34 ± 2	31	32
Mn, mg g DM^{-1}	0.06 ± 0.01	0.06	0.05
Zn, mg g DM^{-1}	0.37 ± 0.8	0.28	0.19
Cu, mg g DM^{-1}	0.050 ± 0.003	0.024	0.016
Cd, mg g DM^{-1}	0.022 ± 0.002	0.017	0.017

* averages ± standard deviations from two different sediment cores

plained by the fact that the top layer contained younger and not yet entirely microbially stabilised organic matter.

Some of the metals, for example Al, Zn, Cu, and Cd, showed a similar pattern as Kjeldahl nitrogen with maximum concentrations in the top layer and lower concentrations in both deeper layers, however, the mechanism of their increase might be rather different. Al started to dissolve from its hydroxyaluminium species present within the soils in the catchment when the base-cation store of soils became depleted due to the increased atmospheric deposition of sulphuric and nitric acids and the consequent decrease of soil pH. Some heavy metals are commonly present in soils precipitated or sorbed on aluminium hydroxides, thus, the mobilisation of Al results also in their mobilisation. In addition to the release from acidified soils, another route of heavy metals into lakes is the direct atmospheric deposition on the lake surface (VESELÝ & al. 1993).

Opposite patterns in comparison with Kjeldahl N and metals were found for total phosphorus and calcium. These elements were less abundant in the top and middle layers but their concentrations were highly increased in the deepest layer. This fact suggests (if our estimate of sediment age is correct) that there had to occur some changes in the catchment of the lake even before the industrial era. A change of forest composition could be one possible explanation. Original undisturbed forests, consisting mainly of beech, fir and spruce, were replaced with spruce-dominated forests during the last centuries. This most likely had an acidification impact on soils and might also resulted in a drop of P and Ca concentrations both in the soils and the water runoff and, consequently, in lake sediments. Another possibility to explain the Ca decrease in the top sediment layer can be its leaching from sediments into the water column under acidified conditions. This process can lead to the depletion and decrease of Ca in older sediment layers than it would correspond with the onset of acidification (VESELÝ, pers. comm.).

The fractionation of total phosphorus in the top sediment layer showed that the predominant part of P (73 %) was combined with metal oxides (Table 6). Only 7 % could be released from the sediment particles in reduction conditions and 18 % was dissolved and/or desorbed with distilled water. The low quantity of P releasable in reduction conditions was in agreement with no observed release of P from the bottom sediments in the lake during anoxia. This result suggests that P in the sediments of Plešné Lake is combined mostly with Al hydroxides and not with Fe hydroxides, from which P would become solubilised during anoxia. The high concentrations of Al and low pH of the water might also contribute to the absence of P release from the sediments by (i) an increased sorption of inorganic forms of phosphorus on hydroxyaluminium species with decreasing pH (DETENBECK & BREZONIK 1991), (ii) an overall decrease of decomposition rates in sediments caused by low pH values (DETENBECK & BREZONIK 1991), and (iii) disabling of enzymatic hydrolysis of organophosphates (OP) by phosphatases due to the formation of complexes OP–Al (JANSSON & al. 1986).

Acknowledgements. The authors thank to E. Stuchlík for the analysis of sulphate and determination of zooplankton, P. Chvojka and I. Krno for determination of benthos, and J. Fott for information about zooplankton. This study was partly supported by GA CR project No. 206/97/0072.

References

ANONYMUS, 1895: Ryby pstruhovité v jezeře Plöckensteinském [Salmonids in Plešné Lake]. *Vesmír 24: 238–239. (in Czech).*

ANONYMUS, 1990: Šumavská putování. Hochficht, Plešné jezero, Třístoličník [Hiking in the Bohemian Forest. Hochficht, Plešné Lake, Třístoličník]. *Rekrea, České Budějovice, 24 pp. (in Czech).*

APHA, 1992: Standard Methods for the Examination of Water and Wastewater, 18th edition. *American Public Health Association, Washington.*

Bendschneider K. & Robinson R. J., 1952: A new spectrophotometric determination of nitrite in sea water. *J. mar. Res. 2: 87–96.*

Buffle J. & Stumm W., 1994: General chemistry of aquatic systems. *In: Buffle J. & De Vitre R. R. (eds.) Chemical and Biological Regulation of Aquatic Systems. Lewis Publ., Boca Raton: 1–42.*

Chábera S., 1975: Přehled vývoje názorů na otázku zalednění Šumavy [An overwiev of opinion development on the glaciation of the Bohemian Forest]. *Šumava 5: 5–7. (in Czech).*

Collos Y., Yin K. & Harrison P., 1992: A note of caution on reduction conditions when using the cadmium–copper column for nitrate determinations in aquatic environments of varying salinities. *Marine Chemistry 38: 325–329.*

Desortová B., 1981: Relationships between chlorophyll-*a* concentration and phytoplankton biomass in several reservoirs in Czechoslovakia. *Int. Rev. ges. Hydrobiol. 66: 153–169.*

Detenbeck N. E. & Brezonik P. L., 1991: Phosphorus sorption by sediments from a soft-water seepage lake. 2. Effects of pH and sediment composition. *Environ. Sci. Technol. 25*: 403–409.

Emteryd O., 1989: Chemical and Physical Analysis of Inorganic Nutrients in Plant, Soil, Water and Air. *Swedish University of Agricultural Sciences, Stencil 10, Umea, 181 pp.*

EPA, 1987: Acid digestion of sediments, sludges and soils – method 3050. *Methods for Chemical Analysis of Water and Wastes. Environmental Protection Agency, Cincinnati, pp. 1–6.*

Fott J., Stuchlík E. & Stuchlíková Z., 1987: Acidification of lakes in Czechoslovakia. *In: Moldan B. & Pačes T. (eds.) Proc. Symp. Geomon 1987, pp. 77–79.*

Frič A., 1873: Über wietere Untersuchungen der Böhmerwaldseen. *Sitz. k. böhm. Ges. Wiss. 3: 103–109.*

Gutzerová N., 1996: Na dno jezera nejen pro poklad [On lake bottom not only for a treasure]. *Šumava (zima ´96/´97): 8–9. (in Czech).*

Hejzlar J. & Kopáček J., 1990: Determination of low chemical oxygen demand values in water by the dichromate semi-micro method. *Analyst 115: 1463–1467.*

Hoppe H.-G., 1993: Use of fluorogenic model substrates for extracellular enzyme activity (EEA) measurement of bacteria. *In: Kemp P. F., Sherr B. F., Sherr. E. B. & Cole J. J. (eds.) Handbook of Methods in Aquatic Microbial Ecology, Lewis Publishers, Boca Raton, pp. 423–431.*

Hruška V., 1986: A simple self-releasing coresampler. *Limnologica, Berlin 17: 259–261.*

Jansson M., Persson G. & Broberg O., 1986: Phosphorus in acidified lakes: The example of Lake Gardsjön, Sweden. *Hydrobiologia 139: 81–96.*

Kopáček J., Hejzlar J., Stuchlík E., Fott J. & Veselý J., 1998: Reversibility of acidification of mountain lakes after reduction in nitrogen and sulfur emissions in Central Europe. *Limnol. Oceanogr. 43: 357–361.*

Kopáček J. & Hejzlar J., 1993: Semi-micro determination of total phosphorus in fresh waters with perchloric acid digestion. *Intern. J. Environ. Anal. Chem. 53: 173–183.*

Kopáček J. & Hejzlar J., 1995: Semi-micro determination of total phosphorus in soils, sediments, and organic materials: A simplified perchloric acid digestion procedure. *Commun. Soils Sci. Plant Anal. 26: 1935–1946.*

Kopáček J. & Procházková L., 1993: Semi-micro determination of ammonia in water by the rubazoic acid method. *Intern. J. Environ. Anal. Chem. 53: 243–248.*

Landa V. & Soldán T., 1989: Rozšíření řádu *Ephemeroptera* v ČSSR s ohledem na kvalitu vody [The distribution of the order *Ephemeroptera* in Czechoslovakia with respect to water quality]. *Studie ČSAV (Praha) 17: 1–170. (in Czech).*

Mackereth F. J. H., Heron J. & Talling J. F., 1989: Water Analysis: Some Revised Methods for Limnologists. *Freshwater Biological Association, Sci. Publ. 36 (Sec. Impress.), 120 pp.*

Münster U., 1991: Extracellular enzyme activity in eutrophic and polyhumic lakes. *In: Chróst R. J. (ed.) Microbial enzymes in aquatic environments, Springer Verlag, New York, pp. 96–122.*

Münster U., Nurminen J., Einiö P. & Overbeck J., 1992: Extracellular enzymes in a small polyhumic lake: origin, distribution and activities. *Hydrobiologia 243/244: 47–59.*

Murphy J. & Riley J. P., 1962: A modified single-solution method for the determination of phosphate in natural waters. *Analyt. Chim. Acta 27: 31–36.*

Novák M., 1968: Údolní nádrž Lipno, geograficko-limnologická studie [A geographic-limnological study of the Lipno Reservoir]. *Práce a studie VÚV Praha, seš. 122: 1–162. (in Czech).*

Papáček M. & Soldán T., 1995: Biogeograficky významné druhy vodního hmyzu (Ephemeroptera, Odonata, Plecoptera, Heteroptera: Nepomorpha) v oblasti Šumavy [The findings of aquatic insects (Ephemeroptera, Odonata, Plecoptera, Heteroptera: Nepomorpha) of the Šumava Mountains important from the physiological point of view]. *Klapalekiana 31: 41–51. (in Czech).*

Porter K.G. & Feig Y.S., 1980: The use of DAPI for identifying and counting aquatic microflora. *Limnol. Oceanogr. 25: 943–948.*

Procházková L., 1960: Einfluss der Nitrate und Nitrite auf die Bestimmung des organischen Stickstoffs und Ammoniums im Wasser. *Arch. Hydrobiol. 56: 179–185.*

Psenner R. & Puczko R, 1988: Phosphorus fractionation: advantages and limits of the method for the study of sediment P origins and interactions. *Arch. Hydrobiol. Beih., Ergebn. Limnol. 30: 43–59.*

Salat J. & Marasé C., 1994: Exponential and linear estimations of grazing on bacteria: effects of changes in the proportion of marked cells. *Marine Ecology – Progress Series 104: 205–209.*

Schmidt R., Arzet K., Facher E., Fott J., Irlweck K., Řeháková Z., Rose N., Straškrabová V. & Veselý J., 1993: Acidification of Bohemian Lakes: Recent Trends and Historical Development. *Bundesministerium für Wisseschaft und Forschung [GZ 45.168/1–27b], 158 pp.*

Schmidt R., Psenner R., Irlweck K. & Meisriemler P., 1991: Umweltanalytische Untersuchungen (Diatomeen, Chrysophyceenzysten, Sedimentchemie) datierter Sedimente des Plöckersteiner Sees zur Gewässerversauerung im östlichen Böhmerwald. *Bericht Amt der O.Ö. Landesregierung, Abt. für Gewässer- und Umweltschutz, 30 pp.*

Sherr E.B. & Sherr B.F., 1993: Protistan grazing rates via uptake of fluorescently labeled prey. *In: Kemp P. F., Sherr B. F., Sherr. E. B. & Cole J. J. (eds.) Handbook of Methods in Aquatic Microbial Ecology, Lewis Publishers, Boca Raton, pp. 695–701.*

Sofron J. & Štěpán J., 1971: Vegetace šumavských karů [The vegetation of corries in the Bohemian Forest]. *Academia, Praha, 56 pp. (in Czech).*

Sommer V., Gliwicz Z. M., Lampert W. & Duncan A., 1986: The PEG-model of seasonal succession of planktonic events in fresh waters. *Arch. Hydrobiol. 106: 433–471.*

Spilka T., 1993: Příspěvek k poznání acidifikovaných jezer na Šumavě [Contribution to a recognition of acidified lakes in the Šumava Mountains]. *Ms. dipl. thesis, Faculty of Sciences, Charles University, Prague, 32 pp. (in Czech).*

Švambera V., 1939: Jezera na české straně Šumavy [Lakes on the Czech side of the Bohemian Forest]. *Sborník české společnosti zeměpisné 45 (192): 15–23. (in Czech).*

Tomšovic P., 1979: Šumavské šídlatky dříve a dnes [The Bohemian Forest ferns in the past and today]. *Živa 27: 122–123. (in Czech).*

TRANVIK L. J., 1992: Allochtonous dissolved organic matter as an energy source for pelagic bacteria and the concept of the microbial loop. *Hydrobiologia 229: 107–114.*

TUČEK F., 1983: Usazování [Sedimentation]. *In: PITTER P. (ed.) Laboratorní metody v technologii vody. SNTL, Praha, pp. 33–53. (in Czech).*

VESELÝ J., 1987a: Okyselení šumavských jezer [Acidification of the Bohemian Forest lakes]. *Vesmír 66676–678. (in Czech).*

VESELÝ J., 1987b: The development of acidification the lakes in Bohemia. *In: B. Moldan & T. PAČES [eds.]: Proc. Symp. Geomon 1987, pp. 80–82.*

VESELÝ J., 1988: Acidifikace šumavských jezer [Acidification of lakes in the Bohemian Forest]. *Ústřední ústav geologický, Praha, 75 pp. (in Czech).*

VESELÝ J., 1994: Investigation of the nature of the Šumava lakes: a review. *Časopis Národního muzea, Řada přírodovědná 163: 103–120.*

VESELÝ J., 1996: Změny složení vod šumavských jezer v letech 1984–1995 [Trends in acid-base status of acidified lakes in Bohemian Forest: 1984–1995]. *Silva Gabreta 1: 129–141. (in Czech).*

VESELÝ J., ALMQUIST-JACOBSON H., MILLER L. M., NORTON S. A., APPELBY P., DIXIT A. S. & SMOL J. P., 1993: The history and impact of air pollution at Čertovo Lake, southwestern Czech Republic. *J. Paleolimnol. 8: 211–231.*

VRBA J., 1992: Seasonal extracellular enzyme activities in decomposition of polymeric organic matter in a reservoir. *Arch. Hydrobiol. Beih., Ergebn. Limnol. 37: 33–42.*

VRBA J., KOPÁČEK J., STRAŠKRABOVÁ V., HEJZLAR J., ŠIMEK K., 1996: Limnological research of acidified lakes in Czech part of the Šumava Mountains. *Silva Gabreta 1: 151–164.*

VYHNÁLEK V., 1994: Determination of chlorophyll a in fresh waters using the whole-water technique. *Journal of Plankton Research 16: 795–808.*

WAGNER P., 1897: Die Seen des Böhmerwaldes. *Duncler et Humbolt, Wiss. Veröff. Ver., Leipzig., 89 pp.*

WENZIG J. & KREJČÍ J., 1860: Der Böhmerwald. *C. Bellmann, Prag, 354 pp.*

WETZEL R. G., 1975: Limnology. *Saunders Comp., Philadelphia, 743 pp.*