CHARLES UNIVERSITY

FACULTY OF SCIENCE

Department of Physical Geography and Geoecology

LUCIE BERANOVÁ

WATER QUALITY AND THE ASSESSMENT OF ANTHROPOGENIC POLLUTION IN THE SEDIMENTS OF THE ELBE RIVER OXBOW LAKES

KVALITA VODY A VYHODNOCENÍ ANTROPOGENNÍHO ZNEČIŠTĚNÍ SEDIMENTŮ FLUVIÁLNÍCH JEZER LABE

Master's thesis

Supervisor: RNDr. Dagmar Chalupová, Ph.D.

Praha 2018

Statement:

I hereby state that I have completed this thesis by myself and that I have properly cited all literature and other information sources I have used. Neither this thesis nor its parts have been submitted to achieve any other academic titles.

Prohlášení:

Prohlašuji, že jsem předkládanou práci zpracovala samostatně a že jsem uvedla všechny použité informační zdroje a literaturu. Tato práce ani její podstatná část nebyla předložena k získání jiného nebo stejného akademického titulu.

In Prague, 20th April 2018
V Praze, dne 20.dubna 2018

Lucie Beranová

Zadání diplomové práce

Název práce

Kvalita vody a vyhodnocení antropogenního znečištění sedimentů fluviálních jezer Labe.

Cíle práce

Cílem práce je zhodnotit kvalitu vody a sedimentů ve fluviálních jezerech Vrť a Kozelská tůň středního Polabí. Součástí práce je vzorkování vody, analýza chemických a fyzikálních parametrů vody a mikroskopická analýza zooplanktonu a fytoplanktonu. Práce zahrnuje také batymetrické mapování Kozelské tůně, odběr sedimentů, jejich zrnitostní analýzu a vyhodnocení antropogenního znečištění subakvatických sedimentů.

Použité pracovní metody, zájmové území, datové zdroje:

Metodická část práce zahrnuje batymetrické měření fluviálního jezera Kozelská tůň metodou River Surveyor, měsíční odběry vzorků vody z jezer Vrť a Kozelská tůň, fyzikální i chemickou analýzu vody v laboratoři Ústavu pro životní prostředí na Přf UK a mikroskopickou analýzu zooplanktonu, řas i sinic na Ústavu technologie vody a prostředí VŠCHT Praha s pomocí Doc. RNDr. Jany Říhové Ambrožové, Ph.D.

Metodická část obsahuje také vzorkování sedimentů, jejich zrnitostní analýzu a vyhodnocení antropogenního znečištění sedimentů na základě stanovení koncentrace těžkých kovů v sedimentech v laboratoři Geologických ústavů Přf UK. Hydrologický režim je hodnocen na základě vodních stavů z nejbližších profilů na řece Labi. Kvalita vody a sedimentů ve starých ramenech je srovnána s údaji z nejbližších monitorovacích stanic na řece, které poskytl ČHMÚ.

Datum zadání: 24.1.2017

Jméno studenta: Bc. Lucie Beranová

Podpis studenta:

Jméno vedoucího práce: RNDr. Dagmar Chalupová, Ph.D.

Podpis vedoucího práce:

Acknowledgements:

I would like to express my very great appreciation to RNDr. Dagmar Chalupová, Ph.D. for her valuable and constructive suggestions during the planning and work on this thesis. Her willingness to give her time so generously has been very much appreciated. I have been extremely lucky to have a supervisor who cared so much about my work, and who answered my questions so promptly.

I also wish to acknowledge the help provided by RNDr. Miroslav Šobr, Ph. D. for his advices and help during the field campaigns. I would like to express my thanks to Doc. RNDr. Jana Říhová Ambrožová, Ph.D. for the work on the microscopical analysis of plankton, and especially her hearty attitude in teaching me all about this interesting field. My gratitude belongs to Doc. RNDr. Zbyněk Engel, Ph.D. for his help with the grain analysis of sediment and to RNDr. Petra Havlíková, Ph.D. for her advices during the assessment of plankton species.

I would like to extend my thanks also to Sylva Nováková for the water samples processing including explanations of determination methods, and as well as to Lenka Jílková and Doc. RNDr. Ladislav Strnad, Ph.D for the chemical analyses of sediments including description of analytical methods, and the opportunity of mercury content determination by myself.

I also thank to Tereza Dlabáčková and Luboš Mrkva for their support during terrain campaigns and especially Petra Stefanescová for her correction of my English.

Special thanks should be given to my husband Lukáš Beran for his incredible patience during teaching me Python software, corrections of my English and mainly everyday support. Finally, I wish to thank to my parents and sister for their support and encouragement throughout my study.

Abstract:

In this thesis, water quality and the assessment of anthropogenic pollution in the sediments of the middle course of the Elbe River oxbow lakes Kozelská and Vrť were studied. It is widely accepted that the oxbow lakes are extremely significant ecosystems. However, a large amount of contaminated material may deposit in these lakes. In addition, the oxbow lakes show the development of the riverbed, and contribute to the stability of the river ecosystem. The research of Lake Kozelská was chosen especially to its proximity to the chemical factory Spolana in Neratovice, which used to be the biggest source of pollution of the Elbe River.

The research included bathymetric measurements, regular observations of hydrological regime, monthly analysis of chemical and physical parameters of water in the period from December 2016 to November 2017, and marginally microscopic analyses of phytoplankton and zooplankton species too. The next part of this research included grain analysis and determination of metal and arsenic concentrations in the sediment fraction of 20µm using Aqua Regia leaching and total decomposition as well.

Concerning water quality assessment, Lake Kozelská and Vrť contained the highest concentration of $N\text{-}NH_4$ among the compared oxbow lakes in the middle course of the Elbe River. Concerning plankton analysis, species occurring in eutrophic reservoirs were found there.

From the point of view of sediment contamination, the highest concentrations of measured elements were determined mainly in Lake Kozelská, which confirmed the hypothesis of the spread of industrial contamination from nearby sources of pollution (Spolana, as in Neratovice) probably also upstream during floods, as reported, for example, in 2002. On the contrary, the sediments of Lake Vrť lake were less contaminated probably due to the impact of the Jizera River, which could dilute the pollution.

Generally, the highest level of contamination was determined in cases of silver and cadmium in the sediments of most of the selected oxbow lakes of the Elbe River. As the research confirmed, at majority of localities in the middle course of the Elbe River, the contaminated sediments of the oxbow lakes represent old anthropogenic loads, which can be remobilized during floods, and this material can represent a secondary source of pollution.

Keywords: water quality, lake sediments, the Elbe river, oxbow lakes

Abstrakt:

Tato práce je zaměřena na zhodnocení kvality vody a antropogenního znečištění sedimentů ve starých ramenech Kozelská tůň a Vrť středního toku Labe. Stará říční ramena jsou významnými ekosystémy, ve kterých se může ukládat velké množství znečištěného materiálu. Tato kontaminace může pocházet z průmyslových zdrojů znečištění především z 2. pol. 20.století. Fluviální jezera také dokladují změny trasy koryta řeky a přispívají ke zvýšení stability říčního ekosystému .Výzkum Kozelské tůně byl zvolen především kvůli poloze tohoto jezera, které se nachází v blízkosti areálu Spolana Neratovice, a.s., která v minulosti představovala největší zdroj labského znečištění.

Tento výzkum zahrnoval batymetrické měření, pravidelné odečítání vodních stavů, měsíční analýzy chemických a fyzikálních parametrů vody v období od prosince 2016 do listopadu 2017. Okrajovou část práce představovala také mikroskopická analýza fytoplanktonu a zooplanktonu.

Další část výzkumu zahrnovala zrnitostní analýza sedimentů a stanovení koncentrace kovů a arsenu v sedimentech ve frakci 20 μm. K výluhu sedimentů byl kromě rozkladu lučavkou královskou použit i celkový rozklad.

Hodnocení kvality povrchové vody v jezerech prokázalo zvýšené koncentrace $N\text{-}NO_3$. Obsah $N\text{-}NH_4$ ve vodě byl v Kozelské tůni i v jezeře Vrť nejvyšší ze všech porovnávaných fluviálních jezer Polabí. Z mikroskopické analýzy planktonu vyplynulo, že v jezerech se nachází převážně druhy, které se běžně vyskytují v eutrofních vodách. Z hlediska kontaminace sedimentů byly nejvyšší koncentrace stanovovaných prvků zjištěny především v Kozelské tůni, což potvrdilo hypotézu o šíření průmyslové kontaminace z blízkých zdrojů znečištění (Spolana, a.s. v Neratovicích) za povodní pravděpodobně i proti proudu řeky, jak bylo zaznamenáno např. za povodně v roce 2002. Naopak sedimenty jezera Vrť byly kontaminovány méně, neboť se zde pravděpodobně uplatnil vliv Jizery, která znečištění zředila. Z hlediska kontaminace sedimentů byla nejvyšší míra znečištění zaznamenána v případě stříbra a kadmia. Jak výzkum prokázal, kontaminované sedimenty fluviálních jezer představují v řadě lokalit v Polabí staré antropogenní zátěže, které mohou být během povodní remobilizovány a kontaminovaný materiál tak může představovat sekundární zdroj znečištění.

Klíčová slova: kvalita vody, jezerní sedimenty, Labe, fluviální jezera

Table of contents

1 Introduction

It is widely accepted that oxbow lakes are extremely significant ecosystems. Besides being the home of a variety of protected species, they increase retention potential and thus play a very important role in flood protection. Oxbow lakes also represent an area in the catchment where a lot of drifted material settles. Therefore, they are also a source of information about historical pollution, which has increased significantly in the Elbe River catchment over the second half of the 20^{th} century. Finally, oxbow lakes show changes in river water course and contribute, in general, to the stability of the river ecosystems.

Unfortunately, the Elbe river has been regulated by people with a system of weirs and dams dating back to the Middle Ages. Its water quality worsened significantly during the second half of the 20^{th} century due to an overuse of fertilizers and a lack of industrial waste water treatment. However, it is still possible to find several protected areas whose preserved oxbow lakes still possess well-functioning ecosystems. In recent years, water and sediment quality of oxbow lakes in the Czech Republic was studies e.g. by (Janský (2005), Chalupová (2007), and Havlíková (2011), who was focused on the development of water quality in Vrť - an oxbow lake that is a part of this study. However, these research works covered only some areas of the Elbe River floodplain. Many oxbow lakes have not been surveyed yet, for instance, Lake Kozelská situated in close proximity to the chemical factory Spolana in Neratovice, which used to be the biggest source of pollution of the Elbe River. In this thesis I have focused on water quality and assessing anthropogenic pollution in the sediments. The research included bathymetric measurements, regular observation of water levels in Lake Kozelská, monthly analysis of chemical and physical parameters of water and analysis of zooplankton and phytoplankton. Grain analysis and determination of metal and arsenic concentrations in sediments were carried out too. The results of the research were compared to previous studies of other oxbow lakes in the region to describe the anthropogenic pollution spread in the Elbe River floodplain.

2 Actual state of research

A significant change in attitude to the environmental issues in the Czech Republic occurred in 1989 after the fall of the “Iron Curtain”. Then, the interest about the quality of the Elbe River began to grow rapidly. Over the next decade, new water treatment technologies have been introduced (Langhammer, 2009). At the beginning of 1990s, the newly established Ministry of the Environment of the Czech Republic launched the national "The Elbe River Project". This project aimed at frequent monitoring to improve the Elbe River water quality, which included also the research of suspended matter and sediments.

Systematic monitoring of water quality in the Czech Republic has been in progress since 1963. The analysis of suspended matter and sediment quality began in 1999. The results from the monitoring are regularly published mainly by the Czech Hydrometeorological Institute and the River Basin Authorities, a state-owned enterprises (SOE).

Research of lakes has almost hundred-year tradition at the Department of Physical Geography and Geoecology at Charles University. First studies were focused on lake bathymetry, then basic physical parameters of water began to be measured (Janský, Šobr, 2003). At the beginning of the 20th century, Šumava lakes were mapped by Švambera in 1939.

The attention of researchers was gradually turning to the water quality studies aimed at chemical parameters of water - e.g. peat bogs acidification research (Oulehle, 2002) or complex limnological studies of oxbow lakes that besides water quality research included sediment analyses too, which was elaborated in the case of diploma theses by Chalupová (2003), Klouček (2002), Šnajdr (2002) and Turek (2004) at the Faculty of Science at Charles University. The issue of water quality and sediments - especially in terms of water erosion and suspended load regime was also studied (Kliment, Matoušková, 2008; Kliment, Langhammer, 2007; Kliment, Kadlec, Langhammer, 2008; Langhammer, Kliment, 2009), ecohydrological monitoring was studied by Matoušková (2005). Non-point sources of pollution in the Elbe River floodplain was investigated by Janský (2002). The classification of lakes and new trends in limnological research were studied by Janský, Šobr and Česák (Česák, Šobr, 2005; Janský, 2005; Janský, Šobr, 2004; Šobr, 2007).

Within the several research projects received at the Department of Physical Geography and Geoecology (e.g. project GAČR “Atlas of the Lakes of the Czech

Republic”), number of studies of various genetic types of lakes were carried out (Oulehle, 2002; Hrdinka, 2004), and several fluvial lakes in the middle course of the Elbe River were studied in detail (Chalupová, 2011; Havlíková, 2011).

Concerning research works carried out at other institutions, Borovec (1995), Tolar and Mráz (1993) were focused on monitoring of metals in sediments in the Czech Republic in the 1990s. The content of selected elements in various sediment fractions was determined by Borovec (2000), who also described the distribution of metals in sediments (Borovec, 2001). Metal and arsenic pollution of sediments and suspended matter was investigated by Lochovsky et al. (1996). Rudiš et al. (2009) assessed the impact of sediment deposition caused by floods of the Elbe River. Bábek et. al (2011) studied anthropogenic pollution in oxbow lakes of the Morava River in detail, which included grain analyzes, dating and geochemical analysis of sediments.

In order to assess sediment contamination, background metal concentrations were determined in the Elbe River floodplain (LichtfuS and Brümer, 1982; Prange et al., 1997). From the point of view of Czech-German cooperation in the Elbe river basin, members of the Department of Physical Geography and Geoecology are regular participants of the Magdeburg seminars on water protection, which were established by the International Commission for the Protection of the Elbe River (MKOL). They also participated in the German project ELSA (Remediation of contaminated Elbe sediments: Schadstoffsanierung Elbsedimente), focused specifically on the hazards associated with the old industrial contamination of sediments in the middle course of the Elbe River and the Bílina floodplain (SedLa project: the Altenmarkt der Elbe and the Seitenstrukturen im Abschnitt von Pardubice / Pradubitz bis Moldaumündung für das Sedimentmanagement im Einzugsgebiet der Elbe) and the lower Czech part of the Elbe River (SedBiLa: Bedeutung der Bílina als historischer und aktuelle Schadstoffquelle für das Sedimentmanagement im Einzugsgebiet der Elbe).

Twelve years of Czech-German cooperation regarding the Elbe River was summed up by Janský (2002). Due to the priority of sediment contamination determination in many countries, the number of studies focusing on this issue has been constantly increasing worldwide. Sediment contamination and monitoring of individual elements in Europe was summed up by Salomon and Brils (2004). Number of studies focused on regional background values determination corresponding to lithological specifics were carried out, however, Index of Geoaccumulation (Igeo) introduced by

Müller (1979) using several categories of sediment contamination have been internationally applied.

At present, there is still a large number of oxbow lakes of the Elbe River, that have not been studied yet, and where the element concentrations of old loads are not known yet.

3 Geomorphological development of the river

Most of Czech rivers spring up in mountainous areas or highlands. The slope decreases with gradually increasing distance from the spring. The kind of geological bedrock, slope of terrain and flow regime of the river influence resulting character of the river valley (Demek, 1987).

The classification of valley was studied by a number of authors. The division to gorge, canyon, V-shaped valley, narrow valley or bed was described by Vitásek (1958). The detailed description was published by Klimaszewski (1978). The four basic types of valley were described by Demek (1987).

In the mountain headspring regions, deep erosion occurs. The streams often creats a valley with bouldery bedrock. Fast flowing water with a significant slope and uneven terrain is highly oxygenated, but it contains a minimum of nutrients. Therefore, water in the headspring regions is very clean. Gradually, the slope of terrain is decreasing, and water velocities also declines (Demek, 1987).

In addition to deep erosion, lateral erosion also occurs. In more flat areas, a river does not flow directly. The bottom is filled with smaller stones and gravel. The oxygenationof water decreases, the nutrient content is increasing, and the river provides better conditions for living to a number of species. If the transition from the mountains to the lowland is too sharp, the river creates "braided" channels with a lot of gravel transported from the mountains (Horník et al., 1986).

Later, the lateral erosion begins to dominate, and the transported material starts to settle more. In the sections of the river, where erosion processes are in balance with sedimentation, the flow is dynamically stable. The subsoil of the riveis composed of fine-grain materials that allow the development of meanders. Due to the amount of nutrients in the water, the primary production increases, however, the organic load can be higher, which could cause the depletion of oxygen during summer. Higher temperature and slow water flow negatively affects the self-cleaning processes and the total diversity of species (Němec, Hladný, 2006).

3.1 Development of river bed

From a long-term point of view, the development of a riverbed is a proof of climate changes that took place in the Holocene period. It also refers to the Earth's crust movements. Rivers have changed their discharge and the ability to transport material due to the fluctuations of precipitation, temperature, or slope changes. River erosion increased due to increased precipitation, melting of glaciers in interglacial periods or tectonic elevations of the riverbasin (Kettner, 1954).

Sand and gravel deposition occurred during dry climate and glacial cooling. It was also influenced by the loss of vegetation, which prevented from surface erosion. In addition, tectonic movements occurred locally again. This all effected the limitation of discharge and the ability to transport material (Kettner, 1954).

The systematic research of the middle course of Elbe River was described by Sokol (1912), Dědina (1918), Engelmann (1938), Žebera (1956), Balatka (1961) and others.

<u>Following river terraces are distinguished (Kettner, 1954):</u>

• *erosion terrace* - it is created by the erosion of a flow and it is covered with a thin layer of sediments

• *accumulation terrace* - it is created by the accumulation process of the flow in a valley

3.2 Formation of meanders

The name "meander" originates from the ancient Greek name of the Maeandros River, now called Mendres, which creates many curves (Pilecká, 1997). Usually, meanders develop in an area with a longitudinal slope of the flow that does not exceed 2%, and a certain width of a floodplain is available (Just et al., 2005). These conditions correspond mostly to the middle and lower course of rivers with the subsoil composed of soft material, which allows easy meandering, when at lower flow velocities in flat areas erosion and accumulation processes are changing. Therefore, there is a greater variability of riverbed shapes. According to the Czech National Standard 736511, a meander is defined as a curve, whose arc angle exceeds 180 °. The steeper bank, where lateral erosion dominates, is called a cut bank, the opposite bank, where the drifted material accumulates, is called a point bar (Figure 1).

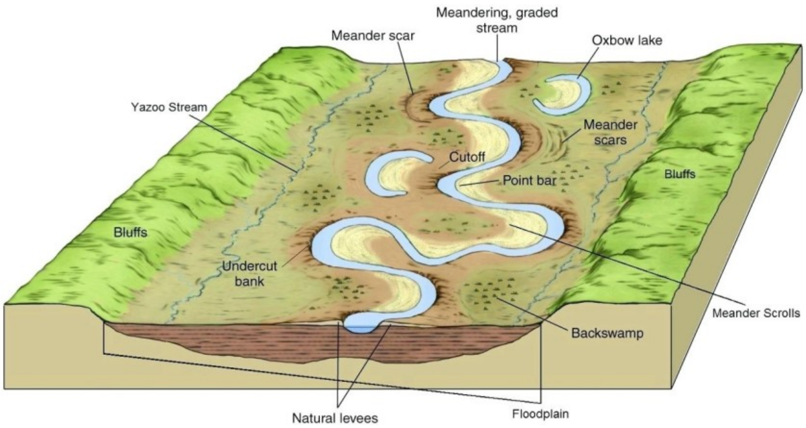

Figure 1:Meandering river diagram (Weiss, 2016)

In the concave area, the streamline directs diagonally to the cut bank and then it turns to the bottom. Pools are created there, which reduces deeper erosion. At the area of inflection, the concave bank changes into convex one, and the river bed gets wider and less deep character. The deposition of drifted material occurs at the point bar area (Just et al., 2005). The dependence of meandering on grain composition or river bank dimension were studied by Schumm (2005). The issue of river meandering according to bedrock and sediment composition was described by Fairbridge et al. (1968).

3.3 Oxbow lakes

As a result of accumulation and erosion processes, straightening the riverbed and artificial cutting of meanders, the oxbow lakes originate. However, floodplain lakes can be also formed by separating a river channel by a river alluvium or by increasing groundwater level during high rainfall or floods (Just et al., 2005).

In the Czech Republic, the human interventions in the riverbeds have been dated from the Middle Ages. With the development of technology, their frequency and scale have grown. The old cut meanders - oxbow lakes - are characterized especially by their elongated and curved shape, which is a proof of the original riverbed (Figure 2).

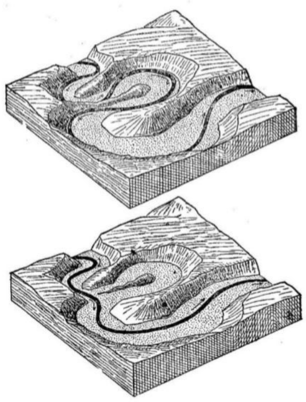

Figure 2: Development of oxbow lakes (Kettner, 1954)

Oxbow lakes usually contain a high amount of nutrients. Due to its position in the fertile floodplains, they are usually also enriched with the material and substances washed out from the surrounding agricultural areas (fertilizers). The supply of nutrients than increases and the lakes often become eutrophicated and silted (Chalupová, 2011).

Concerning the number of oxbow lakes, it is the most represented type of lakes in the Czech Republic, which are very numerous in the middle course of the Elbe River between Hradec Králové and the confluence with the Vltava River (Šobr, 2007).

4 The Elbe River

The management of significant water courses, including the assessment of quality of surface water and groundwater, is carried out by the Elbe River Authority, state-owned enterprise (SOE). Other watercourses e.g. small streams in forests are maintained by the Forests of the Czech Republic, national parks of the Czech Republic, municipalities, or military zone authorities (Němec, Hladný, 2006).

For administrative reasons, the Elbe River is divided into the upper course ending in Opatovice (a few km downstream from Hradec Králové), the middle course extending to the mouth of the Vltava River and the lower course to the state border.

The area managed by the Elbe River Basin Authority covers 14 976 km^2. This area corresponds to 19 % of the area of the Czech Republic. 94% of water from this area flows to the North Sea, 6% flows by the Lužická and Kladská Nisa Rivers into the Baltic Sea (Chalupová, 2011).

4.1 Monitoring of hydrological regime

Water levels of the Elbe River have been recorded since ancient times. However, these old observations were not much systematic. Labels or mars on buildings, bridges or rock walls referred only to individual flood events or droughts (Chalupová, 2011).

At the end of the 1880s, 47 water-level gauges and 446 precipitation stations existed in the Czech Republic. The monitoring of water levels gradually spread into the tributaries, which was also the case of the Elbe River. At the beginning of the 20th century, the water-level gauges were replaced by limnigraphs and discharge was calculated according to the methodology of professor A. R. Harlacher. After World War II, the Hydrological Service was connected with the Meteorological Service, and the Hydrometeorological Institute was established. Currently, the Czech Hydrometeorological Institute operates more than 500 stations (Hladný, 2009). In the area managed by the Elbe River Basin Authority, state-owned enterprise (SOE), almost

120 hydrological profiles exist, where water levels are regularly observed (Chalupová, 2011).

The list of significant floods in recent years is mentioned below:

1997 – July – the Elbe River, the area from Labská to Debrná

2000 – March - the Elbe River, the area from Hostinný – Jaroměř

2002 – August - the Elbe River, the area from Kostelec nad Labem to Hřensko

2006 – March – the whole Czech Republic + floods of the Elbe River in August

2010 – August – the north part of the Czech Republic

2013 – June – the Elbe River and the Vltava River basins

The water level fluctuations during the last 5 years at hydrological profiles Nymburk and Kostelec nad Labem are described in Chapter 8.2 in detail.

4.2 Monitoring of the surface water and sediment quality

Monitoring of water, sediments and biota samples (plankton, benthos, etc.) is carried out by a number of institutions. In 1963, the systematic national water quality monitoring was launched by the Czech Hydrometeorological Institute processing data from external accredited laboratories and maintaining water, biota and sediment quality database.

In the 1990s, the monitoring of water and sediment quality was optimized. An extended monitoring of water and sediment chemistry, and biological indicators were projected in the Elbe River at about 20 monitoring stations. In addition, monitoring of heavy metals and specific organic compounds in sediments were introduced. With some exceptions (e.g. saprobity), a water quality sampling frequency of 12x per a year have been carried out for all indicators. At present, water quality monitoring includes the measurements of oxygen regime, basic and complementary chemical indicators, biological and microbiological indicators, specific organic substances, metal concentrations and radioactivity. At same profiles, benthos is analyzed in order to monitor a long-term contamination with priority pollutants (CHMI, 2016).

In 2016, monitoring of surface water was regularly carried out at 526 monitoring stations (Figure 3), including monitoring of reservoirs resulting from Council Directive 91/676 / EEC on the protection of waters due to pollution caused by nitrates from agricultural sources. Sediment quality was monitored at 62 stations (Figure 4). Water quality was monitored continually throughout the year in Valy, Obříství and Děčín profiles (CHMI, 2016).

Figure 3: Monitoring of surface water quality in the Elbe river basin in 2016 (source: the Elbe River Basin Authority, (SOE))

Figure 4: Monitoring of sediment quality of the Elbe river basin in 2016 (source: the Elbe River Basin Authority, (SOE))

In 2016, a regular monitoring of organic pollutants was extended by new relevant substances, particularly to human and veterinary drugs and advanced pesticides including their metabolites on selected sampling stations. The substances were also included in the watch list according to European Directive 2008/105 / EC (CHMI, 2016).

Besides the systematic monitoring, additional sampling sites were used for the purposes of number of research studies and projects focused on water, plankton, benthos and sediment quality, as it was for examples during the project aimed at the determination of element background concentrations in the Elbe river basin sediments (Prange et al., 1997), or for example in the cases of many limnological studies carried out at the Department of Physical Geography and Geoecology at Charles University (Chalupová, 2011).

Development of water quality of the Elbe River

In 2016, water quality corresponded to class III at majority of sampling stations according to the Czech National Standard 757 211. The headspring areas usually showed the best class of water quality. Class V was found especially at monitoring stations located near big agglomeration with industrial sources of pollution (Figure 5).

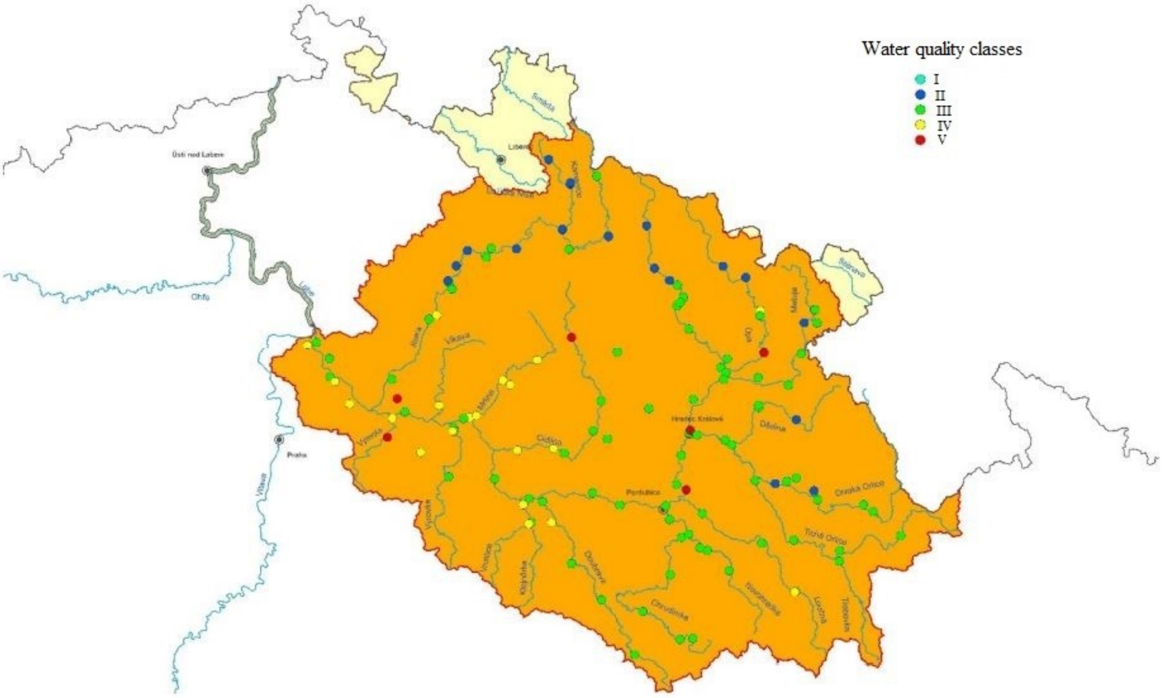

Figure 5: Water quality classes in 2016 according to the Czech National Standard 757 211 (the Elbe River Basin Authority, (SOE))

Water quality in reservoirs could have been affected by the below-average water levels as in 2015. The increased supply of nutrients (especially phosphorus) caused an excessive development of the aquatic bloom of *Cyanobacteria* in some reservoirs, as high spring temperatures were convenient too. (MKOL, 2015).

Long-term development of water quality at monitoring stations in Obříství and Lysá

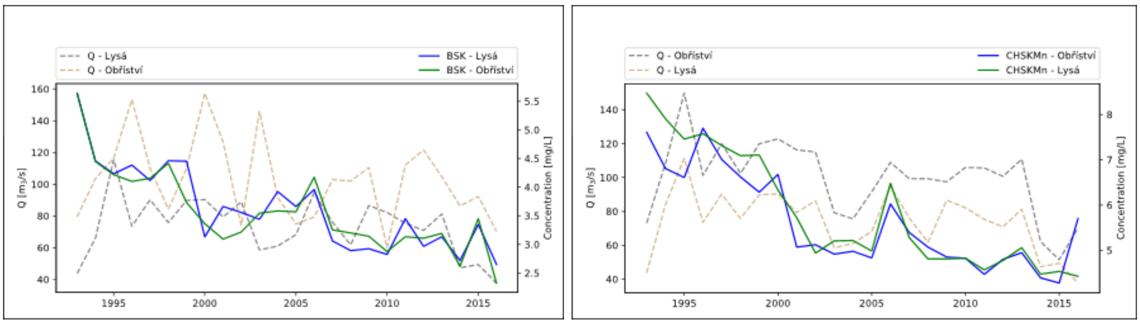

Figure 6: Concentrations of BSK5 and CHSK in Obříství and Lysá nad Labem

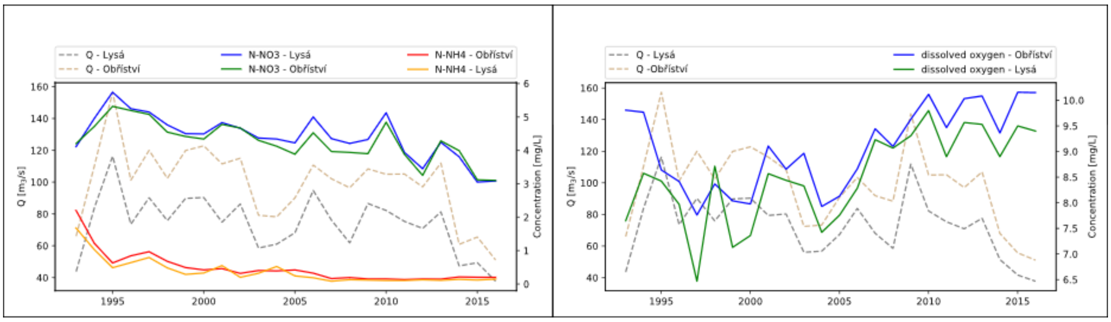

Figure 7: Concentrations of N-NO3, N-NH4 and dissolved oxygenin Obříství and Lysá nad Labem

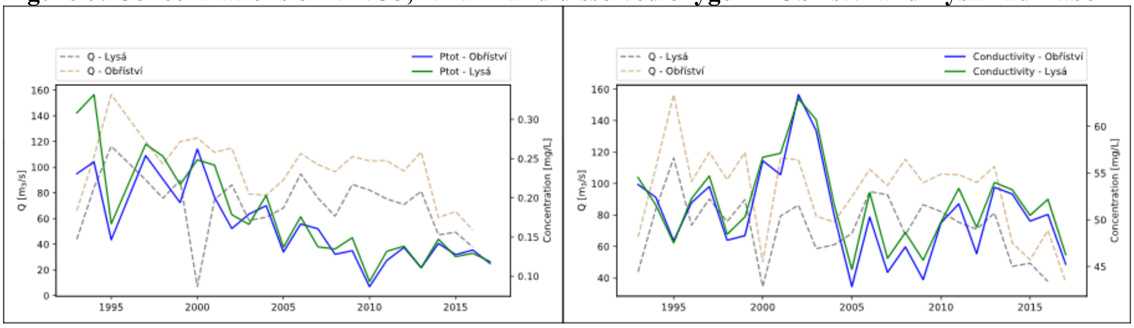

Figure 8: Concentrations of total P and values of conductivity in Obřístv and Lysá nad Labem

At almost every sampling station of the Elbe River, the pollution started to decrease in the 1990s (Figure 6-8). In Lysá nad Labem, the concentrations of almost every measured parameter were lower than at sampling station in Obříství probably due to a greater distance from any source of contamination or dilution with waters from the Jizera River. In contrast, the sampling station in Obříství was located a few kilometers downstream Spolana Neratovice - one of the biggest polluters of the Elbe River. From the above mentioned reasons, water quality in Obříství sampling station showed a higher anthropogenic load (Figure 6-8). However, in the last 20 years, the water quality has improved significantly due to the measures implemented in Spolana and also in other point sources of pollution in the Elbe river basin. The only exception is N-NO_3, whose concentrations has decreased only slightly (Langhammer, 2009). Content of dissolved oxygen gradually increased, which corresponded to the improve of water quality.

Development of water and sediment quality of the Elbe River

At present, the problem of pollution is represented by non-point sources of contamination, which is influenced mainly by land-use in the river basin. In some areas, this kind of pollution got even slightly worse. The situation in smaller watercourses has resulted from local small industries and their limited investments into the waste water treatment. However, the status has improved thanks to state and European subsidies.

The issue of sewage waters from smaller settlements or recreational areas can be also complicated (Langhammer, 2005). Another very problematic issue is the support of agricultural production, which especially in the past meant overuse of fertilizers that have influenced considerably water quality.

The solution for the contamination reduction from non-point sources could be mainly seen in changes in agricultural techniques, land use changes or land modifications. Although the number of inhabitants and point sources of pollution is increasing, the significance of the industrial sector on water quality is declining due to the introduction of modern environmental technologies, whereas the grow of population will continually increase of the amount of municipal wastewaters (Langhammer, 2005).

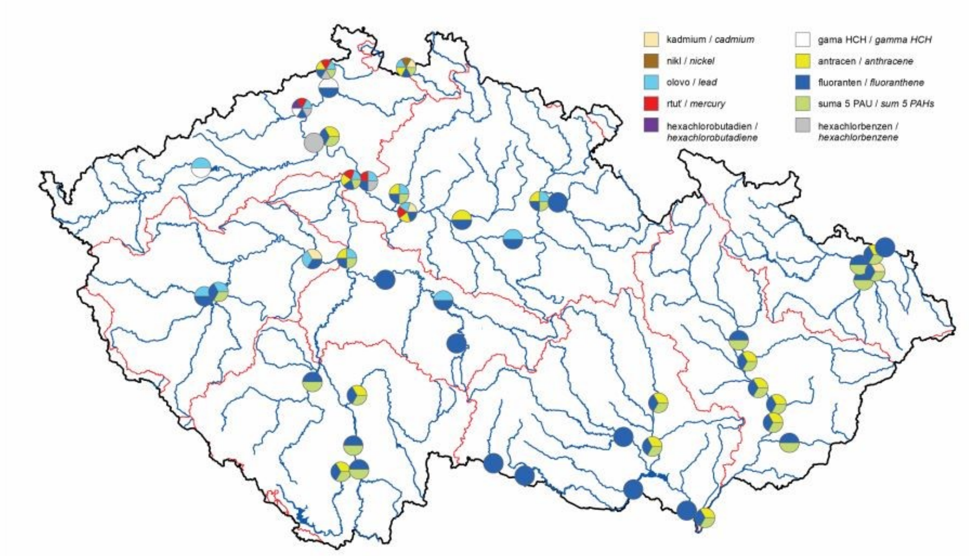

Figure 9: Sediment quality in the Czech Republic in 2015 (Hydrological Yearbook 2015)

Figure 9 shows the sediment contamination according to the concentrations of mercury were high mainly in the middle and lower course of the Elbe River. The impact of industrial plants as Spolana Neratovice, Spolchemie, Synthesia influenced significantly certain part of the basin. The industrial sources of sediment pollution in the Elbe River is discussed in chapter 5.3 in detail.

Development of sediment pollution in Obříství and Lysá nad Labem

Figures 10 and 11 show the development of the concentrations of As, Ag, Cu, Cr, Cd, Pb, Hg, Ni and Zn from 2000 to 2016 at the monitoring stations in Obříství and Lysá nad Labem. In 2004, high concentrations of Cd were measured in Obříství, which could have been caused by the waste water from Spolana with higher cadmium content. In 2014, extreme concentrations of Pb were measured in Lysá nad Labem. Zinc and arsenic concentrations reached almost constant low values in both monitoring stations during the evaluated period.

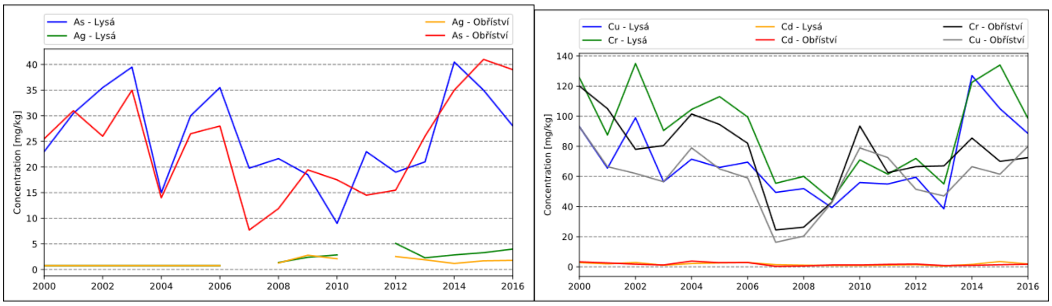

Figure 10: Development of As, Ag, Cu, Cr, and Cd content in sediments in Obříství and Lysá and Labem

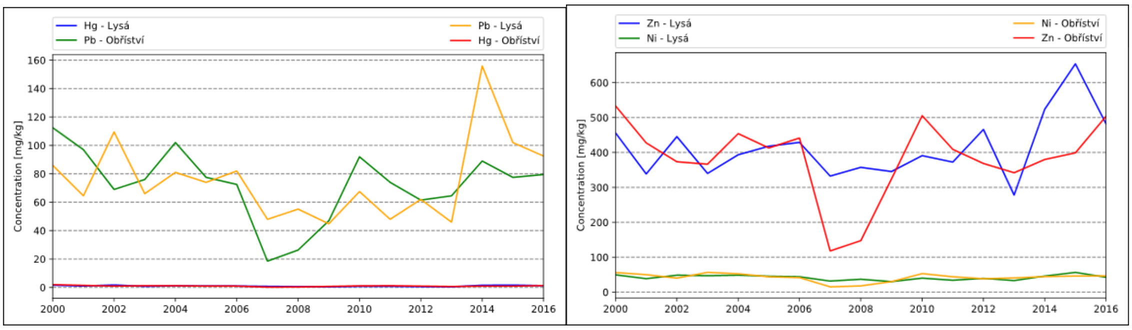

Figure 11: Development of Hg, Pb, Ni and Zn content in sediments in Obříství and Lysá and Labem

<u>The summary of the sediment pollution in the Elbe River in 2015</u>

The content of certain heavy metals, especially zinc, copper and cadmium, almost reached the limit values of pollution in the upper course of the Elbe River. Downstream the industrial plant Synthesia in Pardubice, the significant increase of copper content in sediments was determined. The concentrations of mercury increased in Valy monitoring station, which exceeded the MKOL limit too. In the upper course of the river, the content of mercury was also higher than the limit values. (MKOL, 2015).

The concentrations of zinc exceeded the limit values for the agriculture use of sediment in almost all profiles, respectively cadmium in the half of measuring stations. Concerning indicators of lead, copper and mercury, the limit values for aquatic communities were exceeded in approximately one fifth of 37 monitoring stations. Arsenic concentrations exceeded this limit in one monitoring station only.

The sediments of the lower course of the Elbe River were mostly contaminated with heavy metals, especially cadmium, zinc, copper, lead and mercury. The significant increase of zinc concentrations was measured mainly in Litoměřice. This pollution might be caused by Lovochemie, a.s. Lovosice. The content of mercury in sediments was rising significantly downstream Ústí nad Labem, probably due to the contamination from the Spolchemie - the association for chemical and metallurgical production, and the Bílina River - one of the most industrially polluted tributary of the Elbe River (MKOL, 2015).

4.3 Main sources of pollution of the middle course of the Elbe River

The river floodplain was influenced by human activities since the Middle Ages. The first industrial pollution could come from medieval ore and silver mining in Kutná Hora (Veselý, 1996). After the industrial revolution, the Elbe River floodplain became a strategic center of industry. In the second half of the 19^{th} century, many industrial factories were founded there, and the population and contamination of the river increased. During the straightening of the river, many meanders were cut off. After the World War II, the centrally planned industry without sufficient interest in the environment led to the increase of municipal pollution, the application of chemicals to agriculture, and to the extensive development of chemical industry, usually without appropriate waste water treatment causing a rapid worsening of water quality with its culmination in the 1980s and early 1990s (Janský, 2002; Langhammer, 2004).

Among the main polluters of the Elbe River floodplain to the confluence with the Vltava River were the paper industry in Hostinné, textile, leather and food production in Hradec Králové, photographic industry FOMA in Hradec Králové, power plant Opatovice, Chvaletice and Kolín with several chemical and engineering factories and a refinery. Other sources of pollution were food industry in Nymburk, metallurgical plant in Čelákovice and engineering production in Brandýs. However, the main sources of industrial pollution of the river have been located in Pardubice, where Paramo refinery and tSynthesia chemical plant are located (Trejtnar, 1978).

Synthesia, originally Explosia, was founded for the production of explosives in 1920. Afterwards, a number of inorganic and organic chemicals - pesticides, nitrocellulose derivatives, organic and inorganic acids, solvents, pigments and dyes have been produced here as well.

The second largest chemical plant Spolana Neratovice was founded in 1898. The first production was focused on oils, stearin, soaps, ammonia and gas cleaning. After the World War II, the factory began to develop, ant to produce viscose, chlorine (amalgam electrolysis with the use of Hg), HCl, NaOH, ammonia fertilizers etc.

5 Research sites

To assess the impact of anthropogenic pollution of the Elbe River floodplain, two different oxbow lakes were selected. Both investigated lakes are located in the area between Lysá nad Labem and the confluence of the Elbe River with the Vltava River. The lakes differ in area, depth ratios, the intensity of the hydrological communication with the river and, probably, a way and a source of contamination.

Average annual air temperature of both lakes is 9 °C. Average annual maximum air temperature does not exceed 34 °C. Average annual precipitation varies between 550 – 600 mm and average number of day with snow cover is less than 40 days (TOLASZ, R., et al. Atlas podnebí Česka, 2007).

According to Research Institute for Soil and Water Conservation, this area belongs to Climatic region T2, which is characterized with warm and moderately arid climate, average annual precipitation of 500 – 600 mm, the probability of arid vegetation periods of 20-30 % and average annual air temperature of 8-10°C.

Development of both lakes was very similar. The Third Military Survey from 1852 (Figure 12 and 14) shows that the meanders were still parts of the Elbe River during that time. The meanders were probably cut at the beginning of the 20th century. Current situation is represented by Figure 13 and 15, where the lakes are cut and they are connected only by small canals with the river.

5.1 Lake Kozelská

According to the European administrative mileage effective from 1.10. 2009 (mouth of the Elbe River to the sea is 0 km), Lake Kozelská is located on the right bank of the river between 851,9 and 851,1 km close to Mlékojedy, which is situated in Neratovice district. The lake is connected with the river by small canals at 851,9; 851,6 and 850,1 km. The approximate center of this lake is located at N 50°15.57507', E 14°32.61068'.

This lake is situated on the deluviofluvial clay-sand and claystone sediments. The substrate is made of brown soils on sandy gravel and alluvial soils. This area belongs to the geomorphological classification of subprovince Česká Table, region of Středočeská Table, Staroboleslavská Basin and Dřítská Basin (Balatka, Kalvoda 2006).

Almost every kind of freshwater fish can be found here (pikeperch, carp etc.). Unfortunately, fish are not fully grown due to the premature fishing. Relatively high abundance of coypus lives in this location as well.

During communist regime, adding of permanganate and liming on ice cover of the lake was applied. At this time, the subaquatic sediments were dredged, so the bottom of the lake was cleaned (information from local fisherman).

Canal connecting the lake with the river is surrounded by agriculture fields. The south-east bank of the lake is surrounded by a forest, on the north-west side, agriculture fields (oilseed rape) and pasture for horse-breeding are situated. Weekend cottages are located on the north side of the lake, which may be a local source for pollution as wastewater from households and water from swimming pools are drained into the lake.

Figure 12:The III. Military Survey (www.oldmaps.geolab.cz) and aerial photograph from 1950s of Lake Kozelská (www.cenia.cz)

Figure 13: Current aerial photograph of Lake Kozelská (modified from www.cenia.cz) and current map of Lake Kozelská (www.mapy.cz)

5.2 Vrť

Lake Vrť is located on the left bank of the river between 881,7 and 881,2 river km in Semice, which is situated in Nymburk district. The lake is connected with the river at 881,2 km. The approximate center of the lake is located at N 50°10.13097', E 14°52.02357'.

The geological bedrock of Lake Vrť consists of cretaceous sediments - mostly claystones or clays, opuce and marlite (Chlupáč a kol. 2002). These are covered with pleistocene fluvial sediments, sandy gravels and gravels of river terraces. Near the lake, the floodplain is covered with holocene clays, sandy clays and sandy gravels. According to Balatka and Kalvoda (2006), this location is situated in subprovince Česká Table, region of Středočeská Table, Nymburská basin and Sadská Plain.

Lake Vrť borders with natural protected area Vrť. Present riparian forest shades the part of the lake close to the river. Lake is also surrounded by pine trees and agriculture fields, where vegetable, especially potatoes, onions and cabbage were grown.

Various kinds of fish live in the lake. The most abundant species were carps, pikeperchs, pikes and eels. During communist regime, liming on ice cover was applied in the lake. In the canal connecting the lake and the river, wire mesh was installed in order to prevent the escape of fish from the lake (Havlíková, 2011).

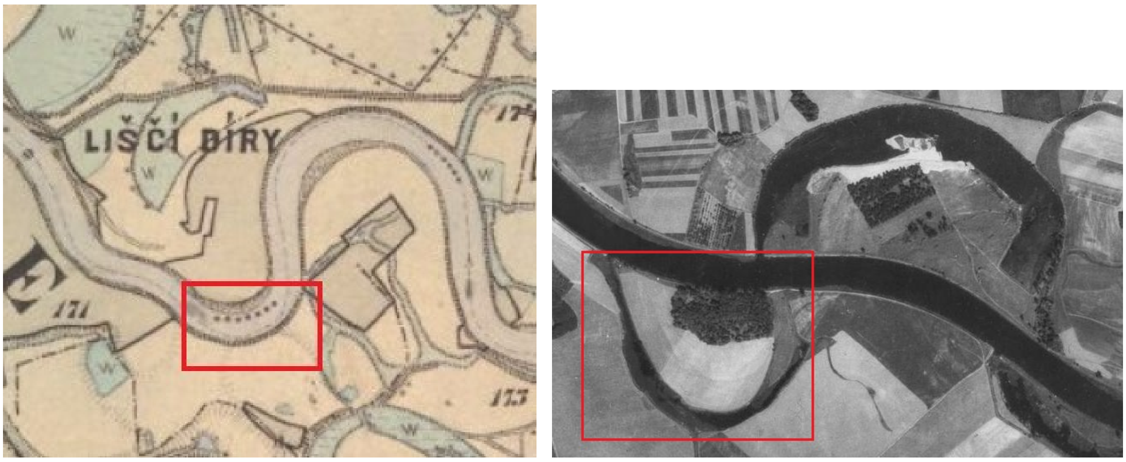

Figure 14: The III. Military Survey (modified from www.oldmaps.geolab.cz) and aerial photograph from 1950s of Vrť (www.cenia.cz)

Figure 15: Current aerial photograph (modified from www.cenia.cz) and current map of Vrť (www.mapy.cz)

6 Material and methods

6.1 Measurement of morphometric characteristics

Morphometric characteristics of Lake Kozelská were measured on 20th April 2017. The bathymetric mapping of Lake Kozelská was done according to the methodology described by Česák and Šobr (2005). The resulting map is available in Appendix 1b. The bank line of the lake was mapped with Garmin GPS using auto-correction by GSM signal (Figure 2). The device RiverSuveyor was used for the measurement of the depths of the lake (Figure 1). This accurate sonar uses Acoustic Doppler Current Profiler (ADCP) technology with the centimeter precision for depth determination. "ADCP uses the Doppler effect of sound waves scattered back from particles within the water column. The ADCP uses the speed of sound in water at the transducer heads to compute depth and velocity" (William, 2001). The measured data was processed in the computer software ArcMap 10.5.

From the measured data, following morphometric characteristics were calculated: lake area (A), lake volume (V), perimeter of lake, lake length (L), maximum width, mean width (w = A/L), maximum depth (Z_{max}), mean depth (Z_{ave}) and depth coefficient ($C = Z_{ave}/Z_{max}$). Values of these parameters are shown in Tab. 1. Moreover, the hypsographic curves were created (Figures 16 and 17).

Parameters of Vrť were already evaluated by P.Havlíková in 2011. Results of morphometric characteristics of both lakes are described in chapter 8.1.

Figure 16: Bathymetric measurement with RiverSurveyor (photo L. Beranová)

Figure 17: Measurement of bank line with GPS Garmin (photo T. Dlabáčková)

6.2 Hydrological regime

To observe the hydrological regime of the lake, water-level gauge was installed in Lake Kozelská (Figure 18). Water levels of this lake were recorded only at the date of sampling water (8th December 2016, 26th January, 9th March, 20th April, 25th May, 10th July, 24th August, 9th October, and 13th November 2017). Zero of the water-level gauge was at altitude of 133,06 m a.s.l. and it was located at E 731483, 289 a N 1025520,193.

Due to the connection of Lake Vrť with the Elbe River by a wide open canal, water levels of Vrť were not observed directly, but derived from the Elbe River water level fluctuations measured at the nearest profile at Nymburk from 8th December 2016 to 13th November 2017. This profile was also used for comparison of the water levels in Vrť and the Elbe River in the research carried out by Havlíková (2011), which confirmed nearly the same fluctuations in the lake. Hydrological river profile in Kostelec nad Labem was chosen for comparison of water levels in the Elbe River and Lake Kozelská. For the long-term assessment of hydrological regime of the Elbe River in both investigated areas, data from 1th January 2013 to 30th December 2017 from Nymburk and Kostelec nad Labem were used.

Figure 18: Water-leve gauge in Lake Kozelská in April and in January (photo D. Chalupová)

6.3 Surface water quality

6.3.1 Characteristics of physical and chemical parameters

Temperature

Temperature is one of the basic parameters that affect physical, chemical and biological processes in water. The main source of heat in lakes is solar radiation, where heat is transferred to deeper layers of water through its flow (Kalff, 2002). Nevertheless, anthropogenic thermal contamination (cooling water mouths, etc.) plays an important role too. In winter and summer, in sufficiently deep lakes temperature stratification occurs. During the period, upper layers of water are separated from deeper, cooler layers by a thermocline - with a rapid temperature decrease (several °C in 1m of depth). On the contrary, in spring and autumn in our latitudes, water of the lakes is mixing throughout the whole water column, which is very important for nutrient supply (Wetzel, 2001). In the case of fluvial lakes, the temperature stratification described above is not usually established due to low depth. The temperature regime of these lakes is highly variable depending on the air temperature and the effects of wind, which can cause mixing of the entire water column during the year.

Alkalinity

The term alkalinity represents the acid neutralizing capacity (ANC) of water to buffer pH changes. It is composed mostly by the sum of weak acid anions and represents a total quantity of base. It is determined by titration of a water sample with strong acid, which neutralizes all hydroxyl, carbonate, and bicarbonate ions (Wetzel, 2001).

Dissolved oxygen (O_2)

Dissolved oxygen in lakes can reveal more about processes in water than measurement of other parameters. Concentrations reflect the actual balance between oxygen supply from the atmosphere, photosynthesis, and metabolic decomposition processes that consume oxygen. Low dissolved oxygen concentrations not only influence biodiversity in aquatic ecosystems, but also have a significant effect on nutrients availability - e.g. phosphorus solubility in water, or pollutant releases from sediments - e.g. toxic trace metals (Kalff, 2002).

Biochemical Oxygen Demand (BOD_5)

Biological oxygen demand represents the amount of biodegradable organic compounds in water. The BOD value is determined by the comparison of dissolved oxygen concentration before and after 5 days of incubation of water sample in the dark. The difference between the two values shows the amount of oxygen needed for the decomposition of organic matter in water sample. Unpolluted waters usually show BOD below 1 mg/L and moderately polluted waters ranges between 2 and 8 mg/L (Kalff, 2002).

Chemical Oxygen Demand (COD_{Mn})

When determining chemical oxygen demand (COD), the concentration of organic matter in water is defined by the amount of oxidizing agent needed on their oxidation. The results are converted to oxygen equivalents and given in mg/L. Oxidation with potassium permanganate was the first invented procedure, however, it is not possible to use it in wastewater analysis. Nowadays, the most used oxidizing agent is potassium dichromate (Pitter, 2015). However, in this research, permanganate was applied for the purpose of comparison with other studies.

Conductivity

Electrical conductivity is a measure of the concentration of charged inorganic and organic substances or ions in water. In dilute solutions, conductivity is a linear function of ion concentration. Electrical conductivity also helps to control the results of chemical analysis of water. From its value, the completeness of the chemical analysis of ionic components of water can be assessed (Pitter, 2015). Conductivity is usually represented by ϒ, which is the inverse resistance of the solution between two $1 m^2$ electrodes spaced 1 m apart. The unit is mS/m. Conductivity depends on ion concentration, charge number and temperature of water. The change of temperature by 1 °C causes a conductivity change of at least 2%. Therefore, it is usually measured at 25 °C or converted to this value (Pitter, 2015).

pH

Acidity of water is caused by the presence of H^+ ions, basicity by OH^- ions. The value of pH is defined as the decimal logarithm of the reciprocal of the hydrogen ion activity (Pitter, 2015). In general, pH values fluctuate from 3 in acidic peat waters to 10 in highly carbonated waters with rich vegetation. In natural waters, intensive photosynthesis associated with the removal of carbon dioxide from water can cause increase of pH values up to 11. On the other hand, low values of pH in acidic waters may release toxic substances e.g. from sediments (Pitter, 2015).

Nitrogen (N)

This biogenic element occurs in waters in several forms. In a molecular form, it diffuses from the atmosphere, however, it is unusable for most organisms. In living organisms, nitrogen is contained in proteins, nucleic acids and other organic compounds. Living organisms excrete nitrogen as ammonia, urea, amino acids and uric acid. Ammonia is also released by the decomposition of dead bodies. With sufficient amount of oxygen, the nitrifying bacteria oxidize it to nitrites and then to nitrates. By denitrification under anaerobic conditions, ammonia and nitrogen gas are created and N_2 can be released back into the atmosphere (Lellák, Kubíček, 1991).

However, this natural cycle is greatly disturbed by many anthropogenic activities. Increased concentrations of ammoniacal nitrogen, which occurs only in minimum concentrations in natural waters, indicate contamination by fecal pollution, chemical industry, communal sewage, food and water industries etc.

Increased contents of nitrates also point to anthropogenic pollution. Decreased concentrations can be found during the vegetation period, when nitrates are consumed by plants. On the contrary, in autumn and winter, their concentrations increase due to the decomposition of organic matter and washing from soils. The main anthropogenic source is agriculture. Both, natural, and artificial nitrogen fertilizers are highly soluble, easily leached, which can contaminate water. However, atmospheric deposition also represents a source of anthropogenic nitrates, particularly in areas with heavy traffic producing nitrogen oxides. Nitrogen toxicity in living organisms is mainly caused by its bacterial reduction to nitrites in the gastrointestinal tract (Pitter, 2015).

Phosphorus (P)

Phosphorus compounds are very important. They can be a limiting factor for production processes or a trigger for eutrophication of natural waters. Soluble forms or phosphorus are primarily orthophosphates and polyphosphates, insoluble are e.g. phospholipids, phosphoproteins, nucleic acids, minerals) (Pitter, 2015). Sediments are the main source of phosphorus, where it is deposited usually in the form of insoluble calcium, magnesium, aluminum and iron phosphates. Soluble orthophosphates, which are already usable by organisms can be gradually released into the aquatic environment from them. (Pitter, 2015). Especially under reducing conditions at the bottom of lakes (summer or winter stagnation), phosphorus is released into water column after reduction of Fe^{III} from insoluble $FePO_4$ (Wetzel, 2001). The phosphorus depletion can occur during peaks of phytoplankton, while high values can be observed in the clear water period (Lellák, Kubíček, 1991). Concerning anthropogenic sources of phosphorus application of soluble phosphate fertilizers influences aquatic ecosystems significantly. Waste waters from laundries, industrial plants or sewage cause contamination of water as well. (Pitter, 2015). Excessive content of this element in water leads to anthropogenic eutrophication characterized by a massive development of algae and cyanobacteria (Lellák, Kubíček, 1991).

Chlorides (Cl^-)

Naturally, chlorides come mainly from minerals. Their concentration in waters corresponds to the geological composition of the site. However, in the investigated lakes, increased concentrations may result from contamination by municipal waste, livestock production or chemical industry. Ice-melting salts are also significant sources of chlorides during winter. Due to the oxidation effects, chlorine is used to purify drinking water.

Calcium (Ca)

Calcium is mainly present in water in the form of ions, exceptionally as suspended $CaCO_3$. The skeletal systems of organisms (fish skeletons, shells, skeletal systems of plants) contain this element as well. Its concentration in water depends on pH, temperature and the presence of other substances, which can be related to conductivity

(Pitter, 2015). Calcium plays also a significant role in bicarbonate buffer system in the aquatic environment. In addition to natural calcium sources, the increased concentration in water mainly comes from industry or the use of limestone or dolomite for neutralization of fishponds. The content of this element is also used for water hardness determination (Pitter, 2015).

Water hardness

This parameter is not uniformly defined through the literature. According to analytical methods, water hardness is represented by the sum of Ca + Mg + Sr + Ba, or Ca + Mg, which is a summary description of water composition. However, it is not precise, because the chemical and biological properties of individual elements are not identical (Pitter, 2015).

Iron (Fe)

The concentration of this element reflects mainly the geological composition of the area (occurrence of ores, composition of soils and sediments). The form of occurrence depends on oxygen ratios, pH and presence of other substances. The bacteria also interfere to the cycle of iron in water. In natural waters, the ratio of ferric and ferrous ions is in equilibrium at standard pH values. Anaerobic conditions cause the reduction to more soluble Fe^{2+} ions. When the amount of oxygen in water is sufficient, $Fe(OH)_3$ is created, which can adsorbs phosphate phosphorus., At the bottom of the lakes, the phosphorus can be stabilized by Fe^{3+} into insoluble form of $Fe(PO_4)_3$ under aerobic conditions, which can lead to a limitation of phytoplankton development. Increased concentrations of iron may come from industrial wastewaters, mainly from ore processing, and metallurgical or chemical industry.

6.3.2 Water sampling

Surface water samples in both lakes were collected on 8th December 2016, 26th January, 9th March, 20th April, 25th May, 10th July, 24th August, 9th October, and 13th November 2017.

The samples were collected from the same sampling site approximately 1 m from the lake banks about 10 cm below water surface. PET bottles and glass samplers for BOD determination were used. During the winter, the surface of both lakes had an ice cover, for that reason, an $1m^2$ holes were cut to make the sampling possible (Figure 19).

Figure 19: Sampling in January 2017 (photo L. Mrkva)

Measurements in Lake Kozelská were made at two different sites to get representative values. The result was calculated as a mean value from both sites. The sampling sites of Lake Vrť and Lake Kozelská are shown in Figure 20.
Samples were usually obtained between 10 a.m. and 2 p.m. and transported to the laboratory before 4 p.m. on the same day.

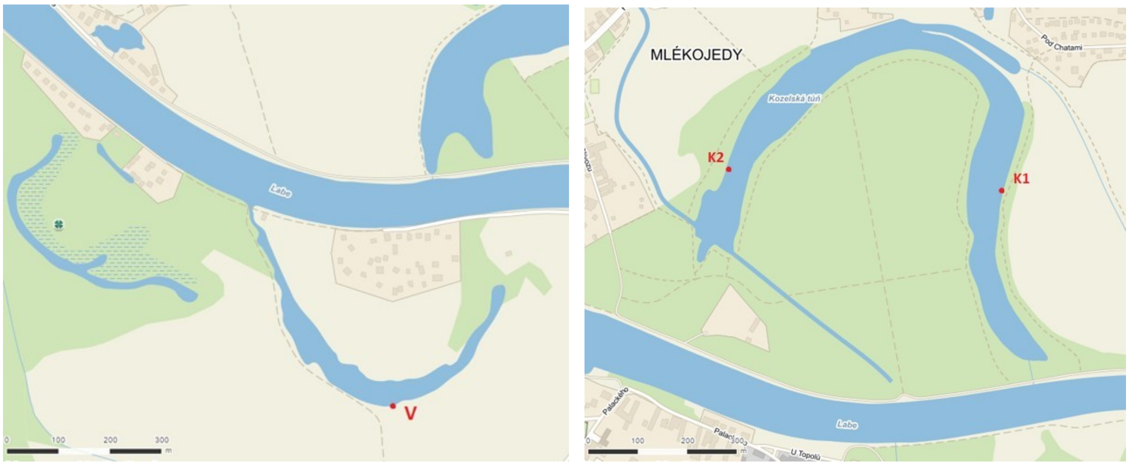

Figure 20: Sampling sites in Lake Vrť (V) and Lake Kozelská (K1, K2) (modified from www.mapy.cz)

6.3.3 Measurement of physical and chemical parameters

Water temperature, dissolved oxygen, pH and conductivity were measured in the field by the multiparametric probe HQ40D Hach-Lange (Figure 21).
Other determined parameters as chemical oxygen demand (COD_{Mn}), biochemical oxygen demand (BOD_5), ammoniacal nitrogen (N-NH4), nitrite nitrogen (N-NO2), nitrate nitrogen (N-NO3), orthophosphate phosphorus (P-PO4), chlorides (Cl^-), alkalinity, calcium (Ca), iron (Fe), manganese (Mn) and water hardness were tested in the laboratory of The Institute for Environmental Studies at Charles University. Conductivity and pH were remeasured in the laboratory as well for comparison with the field data. Laboratory techniques used for determination of parameters listed above are shown in Tab 1.

Figure 21: Field measurement (photo RNDr. D.Chalupová, Ph.D.)

Principles of determination of parameters in detail:

pH

Potentiometric pH is mostly determined by combined electrode method (ČSN ISO 10523 (75 7365)). The principle of the technique is based on the measurement of potential differences between two electrodes immersed in a water sample. One is a reference electrode, which has a permanent and well reproducible potential. The pH values are than obtained from the measurement on the second electrode, which is related to the reference one.

Alkalinity

The total alkalinity is determined by titration of the sample with a 0.1M HCl solution with the use of a mixed indicator or by the potentiometric titration The end of titration is indicated with color change from blue to pink (ČSN EN ISO 9963-1 (757371)).

Chemical oxygen demand - COD_{Mn}

The method is based on the oxidation of organic substances (contained in a sample of water adjusted to a acidic solution using sulfuric acid) with $KMnO_4$. The loss of permanganate (the amount consumed for the oxidation of organic matter) is volumetrically determined. (ČSN EN ISO 8467 (757519)).

Biochemical oxygen demand - BOD_5

After achieving 20 ° C, the water sample is incubated in a fully filled and enclosed glass bottle in darkness for 5 days. Dissolved oxygen concentration is determined before and after the incubation. The volume of oxygen that is needed for a biological oxidation of organic matter is counted from the values *(*ČSN EN 1899-2 (757517)).

Orthophosphates - PO_4^{3-}

The intensity of blue color of orthophosphates after reaction with ammonium molybdate is determined spectrometrically and corresponds to the orthophosphate concentration (ČSN EN ISO 6878 (757 465)).

Chlorides - Cl^-

Concentration of chlorides is determined by the argentometric titration with $AgNO_3$ in the presence of chromate anions. The end of the titration is indicated by color change from yellow to orange-brown. when a slightly soluble Ag_2CrO_4 precipitate is created (ČSN ISO 9297 (757 420)).

Iron - Fe

After dissolution and oxidation to number 3, iron reacts in acidic milieu with thiocyanate, showing a red coloration. The color intensity is spectrometrically determined and corresponds to the concentration (ČSN ISO 6332 (757 433)).

Manganese - Mn

Manganese compounds are quantitatively oxidized by persulfate in nitric acid milieu. The color intensity is proportional to the concentration of manganese and determined spectrometrically (ČSN ISO 6 333 (757 447)).

Calcium - Ca

The concentration of calcium is determined by the titration with EDTA (ethylenediaminetetraacetic acid) in strong alkaline milieu (pH 12-13) to exclude magnesium in the form of $MgOH_2$ with the use of murexid indicator. The point of equivalence is determined by the change in color from pink to violet when the free indicator anion is released (ČSN ISO 6058 (757 416)).

$N-NH_4$

Colorimetric determination is based on the reaction of ammonia with hypochlorite ions and salicylate in alkaline milieu. The resulting substance dissociates to blue color compound (ČSN ISO 7150 - 1).

$N-NO_3$

The concentration of nitrates is measured spectrometrically at 214 nm (UV) directly from water samples. The determination could be interrupted with precipitates of calcium, magnesium, organic compounds or nitrite interference. (ČSN ISO 7890-3 (757453)).

N-NO_2

Sulfanilic acid is diazotized in acidic milieu to diazonium salt. The diazonium salt is coupled with α - naphthylamine to red-violet color. The intensity of the color is proportional to nitrite concentration in the sample (ČSN 830 540 – 11).

Table 1: Laboratory methods of water quality determination (Diode-array spectrophotometer Hewlett Packard 8453 was used to spectrophotometric determination)

parameter	Laboratory method
conductivity	ČSN EN 27888 (757344)
pH	ČSN ISO 10523 (75 7365)
alkalinity	ČSN EN ISO 9963-1 (757371)
water hardness	ČSN EN ISO 11885 (757387)
COD_{Mn}	ČSN EN ISO 8467 (757519)
BOD_5	ČSN EN 1899-2 (757517)
PO_4^{3-}	ČSN EN ISO15681-2 - Flow analysis method (FIA and CFA)
Cl^-	ČSN EN ISO 11885 (757387) - Inductively coupled plasma atomic emission spectrometry (ICP AES).
Fe	ČSN EN ISO 11885 (757387) - Inductively coupled plasma atomic emission spectrometry (ICP AES).
Mn	ČSN EN ISO 11885 (757387) - Inductively coupled plasma atomic emission spectrometry (ICP AES).
Ca	ČSN EN ISO 11885 (757387) - Inductively coupled plasma atomic emission spectrometry (ICP AES).
N-NH_4	ČSN EN ISO 11732 (757454) - Flow analysis method (CFA and FIA) - spectrophotometric detection
N-NO_3	ČSN EN ISO 13395 (757456) - Flow analysis method (CFA and FIA) - spectrophotometric detection
N-NO_2	ČSN EN ISO 13395 (757456) - Flow analysis method (CFA and FIA) - spectrophotometric detection

6.3.4 Characteristics of zooplankton and phytoplankton

Phytoplankton

The distribution and productivity of phytoplankton in relation to environmental and biological factors have received a great attention. It is difficult to predict its distribution due to interactions between species and changes of physical and chemical environment, and by interactions between phytoplankton and its temporally and spatially changing predators as well. The regularity of the biomass cycle and dominant species can be also disturbed by composition change and timing of biomass peaks (Kalff, 2002).

Phytoplankton depends on physical and chemical factors such as light, water column mixing, availability of nutrients, and biological factors, such as competition and predation (SOMMER et al., 1996). It represents the role of primary producers that are consumed by zooplankton and other aquatic invertebrates. In addition, it produces oxygen, which is important for chemical processes and for breathing of heterothrophs

Phytoplankton consists of groups of algae and one significant group of photosynthetic bacteria - *Cyanobacteria* (Hartman, 2005).

The eukarytotic algae form several different phylums. One of them is *Rhodophyta* (red algae), which can be important mostly in oligotrophic streams, where they grow attached to the substrate (Kalff, 2002).

Many species occurring in lakes can be rare in one season, however, when conditions change, they may become frequent. Generally, abundance is not a good measure of growth rate. Some relatively rare small forms grow really fast. In spite of this fact, some show only little changes in abundance because of equally fast predation (Kalff, 2002). However, growing rates are influenced by water column stability, which is characterized by light conditions, nutrient richness, and flushing time (Reynolds, 1997).

Cyanobacteria occur in large colonies, clusters of cells, or single filaments. They often dominate in nutrient rich lakes, and slowly flowing rivers during summer. They can also dominate in ice covered lakes and meltwater streams that freeze in winter. The relative abundance of the large *Cyanobacteria* species is related to the increase of the nutrient concentrations in summer (Kalff, 2002).

Phylum *Chlorophyta* consists of flagellates, cell algae and fibrous algae that have green chloroplasts with chlorophyll *a* or *b* (Hartman, 2005). *Chlorophyta* contribution to green algae is often large in highly polluted polymictic lakes and nutrient-rich farm ponds, as well as in nutrient-rich floodplain lakes and rivers (Happey-Wood, 1988).

Chromophyta is the name for algae with brown chromatophores. Depending on the amount and types of carotenoids, the individual species are distinguished (yellow, brown etc.). Chlorophyll *b* is generally absent. Most species of *Chromophyta* are predominantly found in oligotrophic waters (Hartman, 2005).

Zooplankton

The term zooplankton is used for heterotrophic organisms floating in water. Their active movement is negligible compared to the scale of water reservoirs. They use water movements for their transportation most of the time. Active movement is used only as an escape response in the presence of predators or in prey catching (Lellák and Kubíček, 1991).

Abundance and species composition of zooplankton in water bodies depend on many factors. Kalff (2002) describes zooplankton species composition dependence on water reservoir area - e.g. larger water bodies provide more diverse habitats differing in depth, water temperature, light and oxygen conditions (Havlíková, 2011).

The zooplankton species composition dependence on the trophy was not proven. However, nutrient availability is important for zooplankton biomass. In eutrophic lakes, the biomass of zooplankton is higher than in oligotrophic lakes due to the higher biomass of phytoplankton. Anyway, the timing of zooplankton development with the distribution of phytoplankton, which is significantly affected by the stability of the water column in lakes, is also important (Kalff, 2002).

Shallow water zooplankton as well as deep water zooplankton consist of three major groups of organisms: *Rotatoria, Copepoda* and *Cladocera*. There is also a very small group of *Protozoa*, which is usually not included among zooplankton major groups (Havlíková, 2011).

A relationship between predominant species of *Cladocera* and composition and quantity of fish was found. In the presence of large number of fish in the reservoir, smaller species of *Cladocera* such as *Bosmina longirostris* or *Daphnia cucullata* dominated. After poisoning of fish, the zooplankton composition shifted towards larger species of *Daphnia pulicaria* or *Daphnia longispina*. Furthermore, it was found that changes in the zooplankton community structure depended on the number of fish over fish weight. Thus, a clear relationship between the predominant species body size and the number of fish per unit area of the water reservoir was proven. According to Hrbáček (1962), thismeans that fish are fed with larger species first.

Rotifera are rather larger species living on sediments or macrophytes. Omnivorous representatives (*Keratella, Brachionus, Filinia, Conochilus*) are fed with small picoplankton. Others, such as *Polyarthra* and *Synchaeta*, are selective in choice of food (Kalff, 2002). *Rotifera* are characterized by a very short life cycle. Their eggs develop for one to three days depending on temperature. They mature very quickly and the adults live for one to three weeks. Thus, many generations are born in good conditions and *Rotifera* can significantly increase their abundance in a very short time (Kalff, 2002).

Rhizopoda are protozoans with cilias that are used for movement. They are very variable and live only temporarily. Species consume bacteria, algae and small organisms. Many species live in symbiosis with algae (Hartman, 2005).

Ciliophora are the most advanced protozoa. Their body has a steady shape. In our waters,

there is a great number of *Ciliophora*. They consume small algae and animal remains (Hartman, 2005).

1.1.1.1 Water sampling

Zooplankton and phytoplankton samples were collected from both lakes on 24th August and on 9th October 2017. A plankton net of 40 µm mesh size was used for sampling (Figure 22). Samples were taken in the distance of 2 m from the lake bank by horizontal movements, then fixed with 80% ethanol, and kept in 100 mL bottles. Measurements in Lake Kozelská were made on two different sites to get representative values. The result of species abundance value was calculated as a mean of both measurements. The sampling sites of Lake Vrť and Lake Kozelská are shown in Figure 2.

Figure 22: Plankton sampling in Lake Kozelská (photo RNDr. D.Chalupová, Ph.D.)

1.1.1.2 Microscopical analysis and processing

Samples were observed using the microscope Olympus CX41 with 200x magnification according to the laboratory methods by doc. RNDr. Jana Říhová Ambrožová, Ph.D. (Department of Water Technology and Environmental Engineering UCT Prague) and ČSN 75 7712.

Stirred samples in 10 ml test tubes were centrifugated at 2 000 rev. per minute for duration of 5 minutes. Consequently, the contents of the test tubes were thoroughly stirred with Pasteur pipette. Counting chambers type Cyrus I. with square length of 250 µm were used for plankton determination. The individual of phytoplankton species were determined as a separate cell of maximum size of 100 µm. On the contrary, individuals of zooplankton were counted separately. Number of individuals in 1 ml was calculated according to the formula (1),

$$X = \frac{a \cdot K}{n \cdot z \cdot V} \tag{1}$$

where parameter *a* is the number of individuals in *n* squares, *n* is the number of investigated squares, *z* is the thickening of sample, *K* is the total number of squares in the chamber (Cyrus I: *K* = 1 600), *V* is the volume of chamber in ml (Cyrus I: *V* = 0,01 ml).

In the case of zooplankton, number of species was determined, whereas in the case of phytoplankton, number of phylums was counted. Some species of phytoplankton were determined as well. Unfortunately, due to the difficulty of determination, the exact number of species is not listed. Because of using this semi-quantitative method, the relative abundance of species was calculated. Resulting abundance represents a content of plankton species only in the collected samples, not in the lake.

6.4 Sediments quality

6.4.1 Characteristics of determined metals and arsenic

Silver (Ag)

Naturally, it occurs primarily as argentite (Ag_2S), or forms sulfides with other metals. The dissolved form is mainly represented by Ag^+ and the chlorocomplex. In waste water from the photographic industry, silver occurs in the form of thiosulfatocomplexes, for example $[Ag(S_20_3)_2]^{3-}$ (Pitter, 2015). Simple Ag^+ ion is toxic and dangerous, because it causes brown coloration of skin. Silver is also used for purifying of smaller water sources, its algaecide effects are more effective than in the case of copper (Chalupová, 2011).

Arsenic (As)

In nature, it is mostly found in the form of sulfides. As trace metal it is contained in rocks and soil. Although redox changes are very slow, arsenic in stratified in water reservoirs. It produces volatile methyl derivatives, which are released into the atmosphere. Therefore, detoxification of the environment occurs (Pitter, 2015). It is also significantly accumulated in sediments where it is less stable than mercury. Anthropogenic sources are primarily fossil fuel combustion and ore processing. Arsenic is also present in some pesticides and detergents, which are accompanied by phosphorus (Bencko et al., 1995). Even in small concentrations, it causes chronic poisoning. Toxicity is high for As^{III}, which inhibits biochemical reactions, acts as a nervous poison and has carcinogenic effects as well.

Cadmium (Cd)

Its concentrations in waters are influenced by sediment sorption, especially depending on the content of humic substances. Chlorides do not create volatile derivatives of alkyl, therefore they remain in the environment for a very long time. Significant anthropogenic sources are phosphate fertilizers or sewage water from galvanic plumbing. Cadmium is also present in some pigments and plastic stabilizers. During the combustion of fossil fuels wastes, it gets into the atmosphere from which it is washed by rain (Bencko et al., 1995). Cadmium the second most toxic metal (after mercury), especially its simple ion form (Cd^{2+}). (Bordas, Bourg, 2001). Its inhalation has carcinogenic effects (Pitter, 2015).

Chrome (Cr)

In nature, chrome is represented as chromite mineral ($FeCr_2O_4$) and crocoite ($PbCrO_4$) and also often attend aluminum ores (Merian, et al., 2004). The anthropogenic source is mainly waste water from metallurgy, metal, leather and textile industries. Highest values are recorded in waste water from galvanic plating. Chrome is an essential element (maintains a stable blood glucose level, contributes to nucleic acid synthesis), but at higher concentrations it is toxic due to its carcinogenic and genotoxic effects (Broekaert et al., 1990).

Copper (Cu)

Copper is most often present in the form of sulfides ($CuFeS_2$ - chalcopyrite), bicarbonates ($Cu_3(OH)_2(CO_3)_2$ - azurite) or oxides (Merian, et al., 2004).

The anthropogenic source is mainly waste water from metal processing and a part of copper also comes from atmospheric deposition (Jankowski et al., 2006). It is an essential element for humans. However, copper is very toxic to aquatic organisms (especially Cu^{2+}).

Iron (Fe)

The most common iron ore is pyrite (FeS_2), creel (Fe_2O_3), magnetite (Fe_3O4), limonite ($Fe_2O_3.H_2O$) and siderite ($FeCO_3$). However, these substances are little soluble. Higher concentrations of Fe can be present in sulfuric acid containing oxides originating from pyrite oxidation. Presence of the forms of Fe is influenced by pH or oxidative reduction potential. In water with low oxygen amount, it occurs as Fe^{II} (hypolimnion), while in water with high oxygen concentration, Fe^{III} is present. After mixing water, Fe^{II} is oxidized in epilimnion and solid FeO(OH) is excreted and consequently settled (Pitter, 2015). The reduction can occur at the bottom again. The negative ability of Fe is to color the fabrics, paper, ceramics or food into yellow or brown. It adversely affects the organoleptic properties of water, color and turbidity. Due to possible oxidation and subsequent precipitation on fish gills, limit values are established for carps breeding (Chalupová, 2011).

Mercury (Hg)

Mercury is contained in cinnabar (HgS) and other sulfide ores (Merian, et al., 2004). In waters it occurs as Hg^0, $Hg^{2+,}$ further in the form of hydroxocomplexes and chlorocomplexes. Mercury can also be found as elementary in water. Major anthropogenic sources include ore processing, fossil fuel combustion, industrial waste water, pesticides, or preservatives. Under certain conditions, the source may also include soils and sediments that have long been in contact with contaminated water (Pitter, 2015). It can accumulate in biomass, where it inhibits a number of enzymatic reactions. Mercury has very high bioaccumulation index and it can also damage the liver, kidneys and the brain (Chalupová, 2011).

Manganese (Mn)

Manganese occurs mainly in the form of oxides such as pyrolusite (MnO_2) or braunite (Mn_2O_3). It also flows from soils and sediments. Without oxygen access, the aqueous medium is found to be Mn^{II}. The dissolved form $[Mn(H_2O)_6]^{2+}$ predominates in acid to neutral miele. Under oxic conditions, it is unstable excreted as Mn^{III} and Mn^{IV}. Their composition depends on pH, redox potential, temperature and reaction time (Merian, et al., 2004). The anthropogenic source consists mainly of waste water from ore processing and from metallurgical and chemical industry. Manganese is an essential element for plants and animals but can cause degradation of organoleptic properties of water and brown coloring of materials (Chalupová, 2011).

Nickel (Ni)

Nickel is often found in ores with arsenic and sulfur such as gersdorfite (NiAsS) or NiAs and is also part of some aluminosilicates. Dissolved Ni is represented as $Ni^{2+,}$ hydroxocomplexes, carbonatocomplexes and sulfatocomplexes; Galvanic plating also produces cyano and amine complexes (Pitter, 2015). The anthropogenic source is mainly wastewater from surface treatment of metals or color metallurgy, the glass and ceramics industry. Although nickel is not toxic to humans, it poses a significant risk to aquatic organisms (Bencko et al., 1995).

Lead (Pb)

The occurrence in waters is very dependent on pH and CO_2 concentration. Lead can be present in the acidic mileum as a simple Pb^{2+} ion. An important anthropogenic source are exhaust gases. It releases to water from the lead pipeline, the other source is wastewater from ore processing, color metallurgy, battery manufacture, and the glass industry (Bencko et al., 1995). Lead has a very high bioaccumulation index, and it also significantly accumulated in sediment. It can cause a chronic poisoning and lead is considered as a potential carcinogen (Chalupová, 2011).

Zinc (Zn)

The most common ore is sphalerite (ZnS) and smithsonite (ZnCO3), commonly found in rocks, soil and sediments (Merian, et al., 2004). The source may be atmospheric deposition. It also comes into the water during ore and metals processing, partly released from galvanized objects and is also present in fertilizers (Bencko et al., 1995). For many organisms, including humans, zinc is an essential element as it is part of enzymes and has many biological functions. Diseases of excess are not known, higher water content worses its organoleptic properties. However, for fish and aquatic organisms, zinc is toxic (Chalupová, 2011).

Aluminium (Al)

Aluminum is soluble in water and can be transported from the soil and ground water and then settled in sediments as a vital component for aquatic organisms. Aluminum is very strongly bind to phosphates. This is also why aluminum is effective for water treatment even near the bottom when oxygen is depleted (Klapper, 2002). In water, aluminum occurs in many different forms. At very high pH, the aluminum is again soluble in ionic form. If pH is higher than 5,5, aluminum creates non-toxic complexes. Aluminum is not toxic to fish and aquatic organisms if correct pH range is kept. Therefore, for the successful and safe treatment of the water surface by the aluminum coagulant, it is necessary to maintain the pH in the interval 5.5 - 9 (Klouček, 2005).

Titanium (Ti)

It is a very hard metal occurring in magmatites, especially in the form of oxides - rutile, ilmenite, titanomagnetite. In sediments, titanium is also bound to clay minerals (Petránek, 2007). The main oxidation form is $Fe^{4+.}$ Ti is found bound to other elements in nature and is present in most igneous rocks and in sediments derived from them. Titanium has no biological role. It does not cause poisoning and human body can tolerate titanium in large dose (Lenntech, 2017).

6.4.2 Stabilization and release of metals and arsenic

Under standard physical and chemical conditions in water environment, there is a change in the dissolved phases of toxic metals into solid phases. These are consequently accumulated in sediments (Holz, Pachur, 1992). For this reason, sediments show total load in the area more than the concentration of metallic toxins dissolved in water. Elements released during ion exchange are the most dangerous for the environment (Cd, Cu, Zn). Heavy metals can also be bonded in carbonates (Cd, Zn, Pb, Mn), or on hydrated oxides of Mn and Fe, both as a product of coagulation or adsorption processes. Manganese oxyhydroxides are stable only in strong oxidation, neutral conditions, therefore bound heavy metals can be released very soon after occurring of anoxic conditions. On the other hand, hydrated iron oxides are stable in water under all natural conditions, with the exception of very acidic or reducing medium. Therefore, the release of fixed toxic metals occurs only after long burial of the sediment (Borovec et al., 2000). The most stable bond of toxic metals in the sediment is their incorporation into the organic matter, sulfides, oxides and other crystalline mineral grids (Co, Cr, Ni). This fraction is not available for biota, therefore it represents almost no risk (Borovec et al., 2000)

The finest fraction contains the highest amount of metals due to significantly larger surface of small particles providing many bonds (Chalupová, 2011).

Major immobilization processes include (Pitter, 2015):

- Precipitation - At a higher pH, most metals are precipitated to the form of solid hydrated oxides, carbonates and sulfides. O the contrary, metallic phosphates are precipitated rather in an acidic melie. The reaction also depends on the redox potential, the partial pressure of CO2, the ionic strength of water and the solubility product of the components concerned.

- Oxidation - Higher oxidation stages of most metals are easier to hydrolyze. They exclude less soluble compounds even in a weakly acidic melieu.
- Absorption to the solid phase - Metals are highly accumulated in sediments (clay minerals, hydrated Fe and Mn oxides or organic matter).
- Incorporation to biomass - Metal adsorption occurs more often than the active transport to cell.

On the contrary, major remobilization processes are (Pitter, 1999; Mrňa, 1991):

- Dissolving - Stable solid metal compounds are dissolved at decrease of pH, change of redox conditions or the effect of natural and synthetic solvents.
- Reduction - Slightly soluble metal compounds are more soluble in reduced form than similar compounds in oxidized form. However, sulfides reduced to sulfides immobilize metals under anaerobic conditions.
- Complexation - Natural or anthropogenic inorganic and organic complexing agents prevent from secretion of poorly soluble compounds (hydroxocomplexes, humic acid carbonates, amino acids, bacterial extracellular polymers). Stability of complexes depends on cation wich creates complexes (complexes with Pb^{2+} are more stable than complexes with $Cd^{2+)}$ and pH (stability increases with increasing pH). Very stable complexes are formed at cyanide, ammonia and EDTA and NTA at higher pH.
- Desorption - Metals bound to low soluble substance are released at pH decrease or during redox potential change. Desorption occurs after reduction of hydrated Fe and Mn oxides, or when pH is decreased due to sudden aeration (acid production and H^+).
- Increase of the content of certain substances in water - Higher salt content leads to unsuccessful metal competing for bonds and their subsequent release. The dissolution of the bonds occurs due to contamination with anthropogenic substances inducing changes of pH or redox potential.
- Release from dead biomass - Metals are released during decomposition processes.

6.4.3 Sediments sampling and processing

Sampling of sediments of both oxbow lakes was realized on the 5th September 2017. Sediment cores were obtained with a "Beeker Sampler" Eijkelkamp from a boat (Figure 23). This sampler is suitable for sampling less cohesive layers of sediments below water surface. Originally, it was planned to sample from two different sites of Lake Kozelská, however, this was not possible because of the presence of sandy layers not suitable for the determination of metal concentrations. Sites evaluated as convenient for sampling are shown in Figure 7. GPS coordinates of the sampling site in Lake Kozelská are N 50°15.26722', E 14°32.71882' and coordinates of the site in Vrť are N 50°10.13468', E 14°52.01520' (Figure 25). The lake sediment cores were sliced into subsamples of 10 cm thickness in order to analyze each layer separately (Table 2). Sediment core in a plastic tube from Lake Kozelská is shown in Figure 24. Individual samples were put to the hermetic ziplock bags and stored in a fridge.

Table 2: Sediment cores separation into the individual layers

depth [cm]	Vrť	Lake Kozelská
0 - 7	x	x
7 - 17	x	x
17 - 27	x	x
27 - 37	x	x
37 - 47	x	x
47 - 57		x
47 - 59	x	

Figure 23: Sediment sampling from a boat (photo T.Dlabáčková)

Figure 24: Sediment core (photo L.Beranová)

Figure 25: Sediment sampling site in Vrť (V) and Lake Kozelská (K)

6.4.4 Methods of determination of metals and arsenic

Due to relatively low background concentrations of heavy metals and As in the middle course of the Elbe River, their increased content in the sediments of researched oxbow lakes indicates anthropogenic contamination. For that reasons, the concentrations of Ag, Al, As, Cd, Cr, Cu, Fe, Hg, Ni, Pb, Zn, Ti were determined in the lake sediment profiles.

<u>Methods of determination</u>

For comparability of results, it was necessary to use the same sediment grain fraction that had been used in previous research works. The use of 20µm was based on conclusions of the study by F. Ackermann (1983), research works carried out by Prange et al. (1997), and other researchers (Chalupová, 2011). Moreover, the fine fraction is also used in the systematic monitoring of the Elbe River sediments carried out by the Elbe River Basin, a state-owned enterprise (SOE), which makes the results comparable. The wet sieving (20μm stainless sieve) was conducted in the Laboratory of Physical Geography at Charles University.

Heavy metal and arsenic contents in sediment samples were obtained with the use of the aqua regia leaching method referred as "pseudo total digestion" (Figure 7), and total digestion as well (7.4.4).

The difference between these two methods is that the concentrations of heavy metals and arsenic obtained by pseudo-total digestion do not include the amount of analyzed element that is contained in silicate fraction in the sediment. Therefore, the resulting metal concentration represents approximately the amount, which may be bioavailable. On the other hand, total digestion with very strong acids represents not only anthropogenic enrichment, but also total natural content of an element in the sediment.

The sediment analysis of the samples from Lake Vrť included only aqua regia leaching, in case of Lake Kozelská, both methods were applied.

<u>Aqua regia leaching process</u>

Aqua Regia (2,5 ml of HNO_3 + 7,5 ml of HCl) was poured over 0,5 g sample in Savilex pressure vessels. These were stored at room temperature overnight, then heated to 105 ° C for 6 hours. After cooling, the solution was transferred to the volume of 50 ml. For ICP OES measurements, 10x and 100x dilutions were used.

<u>Total decomposition process:</u>

A 0,2 g sample was ignited at 550 ° C. Afterwards, 10 ml of HF and 1 ml of $HClO_4$ were added and heated on a sand bath to the production of white smoke. Consequently, 5 ml of HF and 1 ml of $HClO_4$ were added and fumed again. The residue with water was than heated with 2 ml of HNO_3 to 105 ° C to dissolve. After cooling, the solution was transferred to a volume of 100 ml. Undiluted and 100x diluted solutions were used for ICP OES measurements.

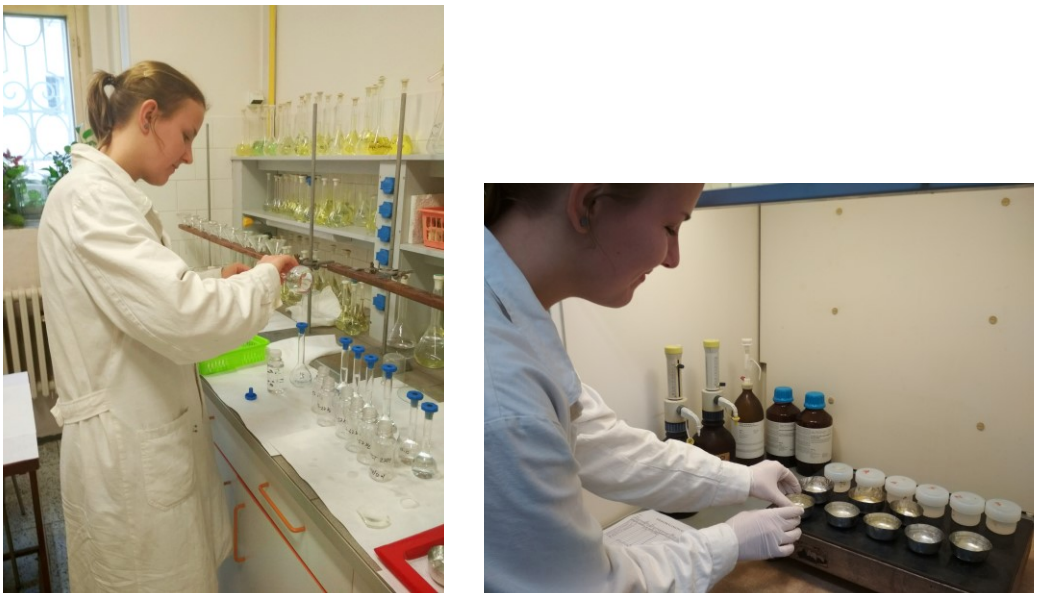

Figure 26: aqua regia leaching (photo L.Jílková)

The concentrations of investigated elements were determined by the inductively coupled plasma optical emission spectrometry (ICP – OES). “It is a type of emission spectroscopy that uses the inductively coupled plasma to produce excited atoms and ions that emit electromagnetic radiation at wavelengths characteristic of a particular element. It is a flame technique with a flame temperature in a range from 6000 to 10000 K. The intensity of this emission is indicative of the concentration of the element within the sample” (Heiftie, 1982). The analysis was conducted in the Laboratories of the Geological Institutes at Charles University.

Mercury content (Hg) determination

The concentration of mercury was determined using the Atomic Absorption Spectrometer AMA-254 (Figure 27), which allows spectral determination from liquid and solid samples without prior treatment. “The spectrometer uses the principle of generating metallic mercury vapor by thermal decomposition of the sample in the combustion tube. The vapor is then captured and concentrated on a gold amalgamator. This principle achieves high sensitivity without dependence on the matrix. A minimum sample size is sufficient for determination and the detection limit is in the hundredths of ng” (Száková *et al.*, 2004). The analysis was conducted in the Laboratories of the Geological Institutes at Charles University as well.

Figure 27: AMA-254 Spectrometer (photo L. Beranová)

6.4.5 Methods of Grain analysis

Not only sediment composition, but also particle size, are decisive in many physical and chemical processes occurring in sediments. For example, the concentration of heavy metals in sandy fractions is lower than in silt. The grain composition also shows the changes of erosion-sedimentation processes and indicates the speed and the way of settling of material in oxbow lakes and the floodplain. The composition of the sediments represents also a historical record of processes occurring in the river-bed including changes of river bed position (Chalupová, 2011).

As fine grained sediment samples were collected (diameter of particles < 2 mm), laser granulometry could be applied. Samples preparation was as follows: present organic matter had to be removed from the sample with 30% solution of hydrogen peroxide. It was also necessary to eliminate the calcium with 3% HCl solution as well. Subsequently, the presence of calcium in the sample was tested with ammonium oxalate.

After each application of a solutions, the samples were mixed with the use of magnetic stirrers (Figure 28), and then centrifugated at 2 700 rev./min for a duration of 2 minutes. In the end of the preparatory treatment, $(NaPO_3)_6$ was used as a fixator of the sample. Afterwards the samples were analyzed in a laser granulometer.

Figure 28: Magnetic stirrer (photo: L. Beranová)

Laser granulometer works on the principle of a laser diffraction analysis. It is a technology that uses diffraction patterns of a laser beam passed through sample to measure geometrical dimensions of particles (de Boer et al., 2003). “Laser diffraction analysis is based on the Fraunhofer diffraction theory, defining that the intensity of light scattered by a particle is directly proportional to the particle size. The angle of the laser beam and particle size have an inversely proportional relationship, where the laser beam angle increases as particle size decreases and vice versa” (McCave, 2013). These analyses were performed in the Laboratory of Physical Geography at Charles University.

7 Results of the research

7.1 Morphometric characteristics

Morphometric characteristics of both lakes are shown in Table 3.
According to the bathymetric map of Lake Vrť (Appendix 1a), the maximum depth of water was located in the middle part of the lake. Settling of sediments was obvious in both ends of the lake. In the western part near the canal connecting the lake and the Elbe River, intensive deposition of sediments occured at elevated water level period. In the eastern part, the depth of sediments was increased due to vegetation, especially reed (Havlíková, 2011).

As the bathymetric map of Lake Kozelská shows (Appendix 1b), the greatest depth of the lake was located at the cut-bank, where bank disturbance and erosion occurred, which was caused by the higher speed of water in the original meander. Therefore, lake banks are very steep in this area. The smallest depth was, on the other hand, at the sand point bar, where the speed of water was slower. At some bank parts of the lake, the depth measurement was not possible due to reed. These places are crosshatched on the map. Most of the eastern parts of the lake ware deeper than the western areas.

Table 3: Calculated morphometric characteristics

Parameter	Vrť	Lake Kozelská
lake area (A) [m²]	25950	122584
lake volume (V) [m³]	14556	125010
perimeter of lake [m]	1778	3562
length of lake (L) [m]	821	1509
max width [m]	61	100
mean width [m]	32	81
max depth Z_{max} [m]	1,6	2,47
mean depth Z_{ave} [m]	0,56	1,02
depth coefficient C	0,35	0,41

Lake Kozelská with its area (122 584 m^2) and volume (125 010 m^3) was above average among oxbow lakes. On the contrary, the area (25,950 m^2) and the volume (14,556 m 3) of Lake Vrť belonged to medium size oxbow lakes. The maximum depth of Lake Kozelská was 2,47 m, whereas the maximum depth of Vrť was 1,6 m. Every morphometric characteristic of Lake Kozelská was higher than the parameters of Vrť.

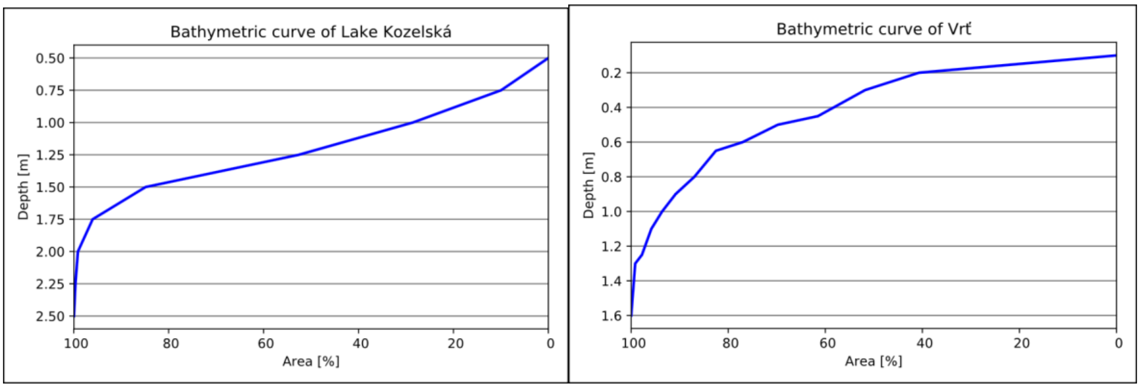

Figure 29: Bathymetric curve of Lake Kozelská and Vrť

Bathymetric curves (Figure 29) show detailed illustration of the depth ratios of the oxbow lakes. During communist regime, sediments of Lake Kozelská were dredged, which could affect the natural shape of the lake bottom (information from a local fisherman). Moreover, the banks of Lake Kozelská were steeper than the banks of Vrť.

7.2 Hydrological regime

Figure 30 shows water level fluctuations in Lake Kozelská and Kostelec nad Labem during measured period. As it can be seen from this picture, water levels of the lake corresponded in most cases to changes in water levels in the river, which can prove hydrological communication between the lake and the river. Zero of the water-level gauge in Lake Kozelská was located at altitude of 133,06 m a.s.l. At the hydrological profile in Kostelec nad Labem, water level fluctuations of the Elbe River were measured, and the zero of water-level gauge was located at 157,83 m a.s.l. there.

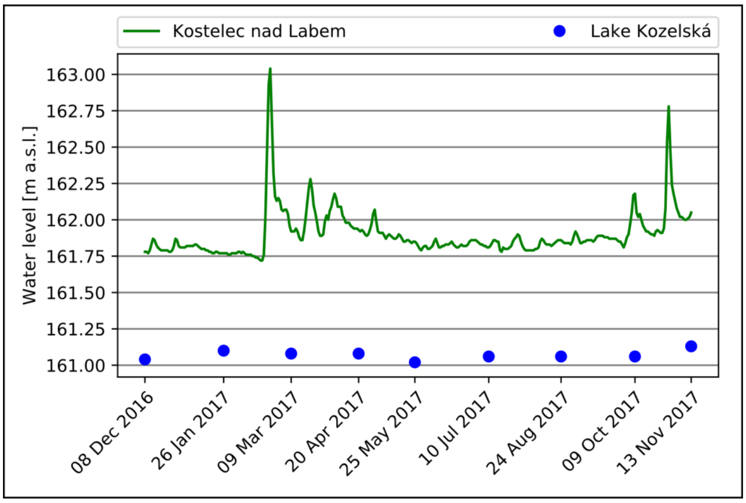

Figure 30: Water level fluctuations in Lake Kozelská and Kostelec nad Labem (Zero of the water-level gauge in Lake Kozelská located at altitude of 133,06 m a.s.l. and in Kostelec nad Labem at 157,83 m a.s.l.)

The highest water level of the Elbe River was recorded in February 2017, which could be caused by melting of snow. From March to April, water levels were still relatively high. On the contrary, low water levels in the river were observed during summer, which corresponds to relatively low water level values in Lake Kozelská. The decline in water levels might be caused by higher temperature and less precipitation. Water levels started to rise in September and October 2017. According to the report of the Czech Hydrometeorological Institute, precipitation was above-average - precipitation of 78 mm represented 182 % of normal average value measured in October 2017.

Table 4 shows water level fluctuations in Lake Kozelská, and also in the profiles of the Elbe River - Kostelec nad Labem and Nymburk at the dates of sampling. The biggest differences in water levels were observed in Kostelec nad Labem, while in Lake Kozelská, minimum changes in water level were recorded.

Table 4: Water level fluctuations in Lake Kozelská, Kostelec nad Labem and Nymburk. (Zero of the water-level gauge in Lake Kozelská located at 133,06 m a.s.l., in Kostelec nad Labem at 157,83 m a.s.l. and in Nymburk at 180,7 m a.s.l.)

	Water levels at the time of sampling [m a.s.l.]		
Date	**Kostelec nad Labem**	**Lake Kozelská**	**Nymburk**
08.12.2016	161,78	161,04	182,61
26.01.2017	161,77	161,1	182,57
09.03.2017	161,92	161,08	182,66
20.04.2017	161,93	161,08	182,61
25.05.2017	161,85	161,02	182,66
10.07.2017	161,81	161,06	182,66
24.08.2017	161,86	161,06	182,66
09.10.2017	162,18	161,06	182,7
13.11.2017	162,05	161,13	182,69
max.	162,18	161,13	182,70
min.	161,77	161,02	182,57
difference	0,41	0,11	0,13
mean	161,91	161,07	182,65

Figure 31 shows the water level changes in Nymburk. Considering Vrť is hydrologically connected with the Elbe River by wide surface canal, values in Nymburk profile should correspond to the water level fluctuations in Vrť. Distribution of these values during the year show a similar trend as in Lake Kozelská. The lowest values were observed in December, while from January to April, the highest water levels occurred. This might be caused by melting of snow during spring. During the whole summer, water levels were constantly lower. They started to rise again in October, when extreme precipitation was recorded (the Czech Hydrometeorological Institute).
Zero of the water-level gauge was located at 180,7 m a.s.l. in Nymburk.

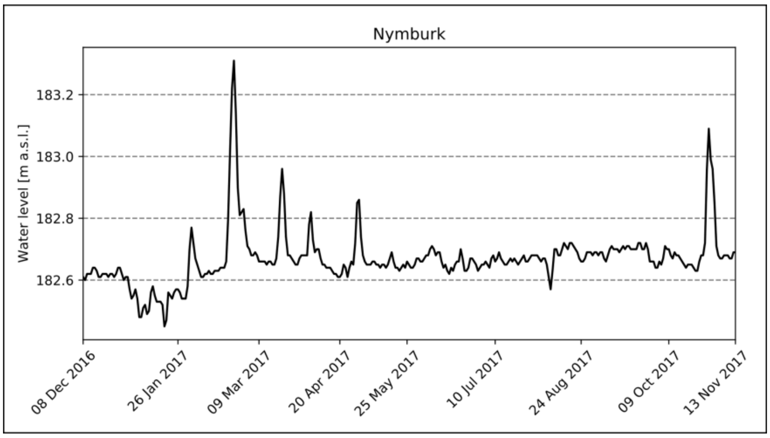

Figure 31: Water level fluctuations in Nymburk with marked dates of sampling in Vrť (zero of water-depth gauge located at 180,7 m a.s.l.)

Long-term changes of water levels in the Elbe River from 2013-2017 from the both above mentioned hydrological profiles are shown in Figure 32 and 33. The water level fluctuations were very similar in Nymburk and in Kostelec nad Labem. Therefore, following description can be related to both profiles. Water levels reached extreme values in June 2013, when almost 100-year flood occurred. On the contrary, in spring 2014, drought affected the river basin, therefore, very low water level values were recorded during that time. At the beginning of year 2015, high values were observed, probably due to melting of snow. In contrast, in July 2015, drought occurred again and the lowest water levels from 1952 were measured. Generally, at the beginning of almost every year, water level rose due to snow melting and the lowest water levels were usually measured in summer because of higher temperature and less precipitation.

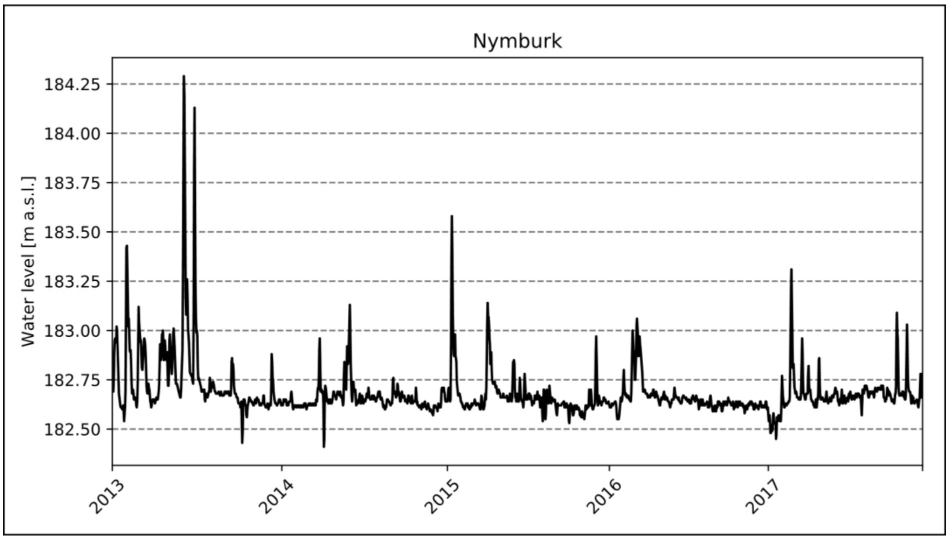

Figure 32: Long-term water level fluctuations from 2013 - 2017 in Nymburk

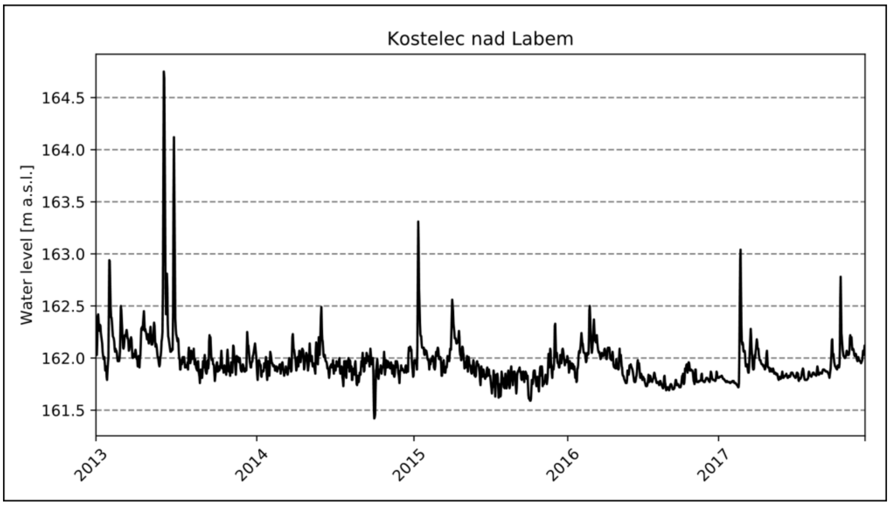

Figure 33: Long-term water level fluctuations from 2013 - 2017 in Kostelec nad Labem

7.3 Surface water quality

Physical and chemical parameters of water have an important impact on water ecosystem in determining the structure of biotic communities and productivity of aquatic systems. These parameters also influence transport, release and biological availability of toxic metals in subaquatic sediments.

7.3.1 Temperature, pH, alkalinity and conductivity

<u>*Surface water temperature*</u>

Measured temperatures represented a typical seasonal distribution during the year (Figure 34). Values of both lakes showed very similar pattern. In July, temperature reached up to 27,35 °C in Lake Vrť and 26,8 °C in Lake Kozelská. In January, temperature dropped down to 0,4°C in Vrť and to 0,69 °C in Lake Kozelská. During the winter the surface of both lakes had ice cover. The thickness of the ice was 16 cm in Vrť and 11 cm in Lake Kozelská that January.

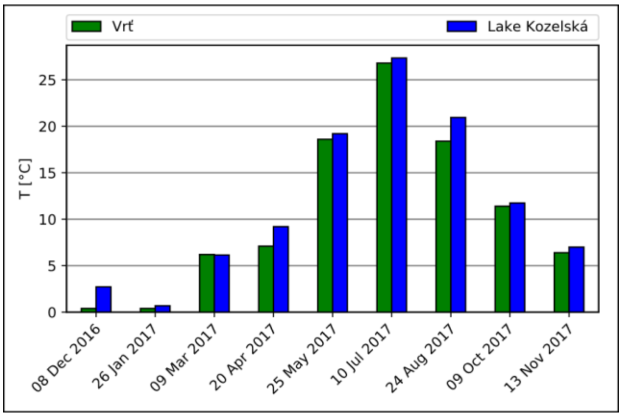

Figure 34: Temperature in Lake Vrť and Lake Kozelská

<u>*pH*</u>

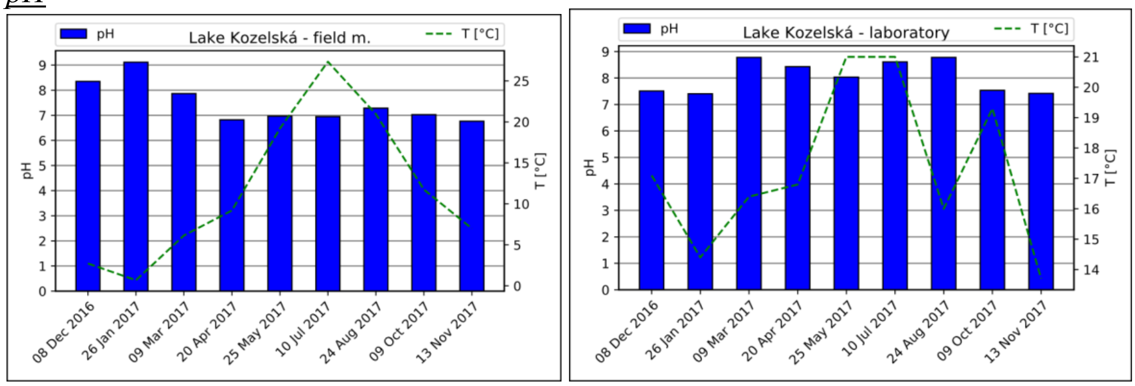

Figure 35: pH of Lake Kozelská measured in the field and in the laboratory

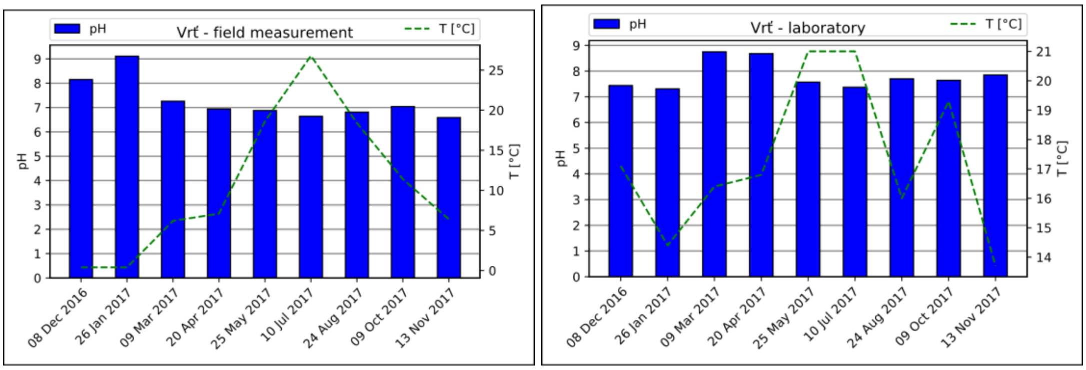

Figure 36: pH of Lake Vrť measured in the field and in the laboratory

As Figures 35 and 36 show, pH ranged in interval 6,5-9,11. Most of the values represent neutral or slightly alkaline environment. Results from both, field measurements and laboratory analysis, were obtained. Generally, pH values measured in laboratory weres higher. That applies especially in summer. This could have been caused by changing of conditions during the transport of samples or during laboratory analysis when another pH meter was applied.

Slight increase of pH values was measured at the end of winter and at the beginning of spring. The change was driven by the increase of phytoplankton activity that causes CO_2 exhaustion from water during this period.

In the summer, values in Vrť were higher than in Lake Kozelská. This could correspond with significantly higher amounts of dissolved oxygen in Vrť during these months. The field measurements showed a negative relation between pH and water temperature.

Alkalinity

Alkalinity is described by the means of acid neutralizing capacity (ANC). Higher ANC means decrease the change in pH after addition of acid or base, meaning the water is more buffered (Kalff, 2002).

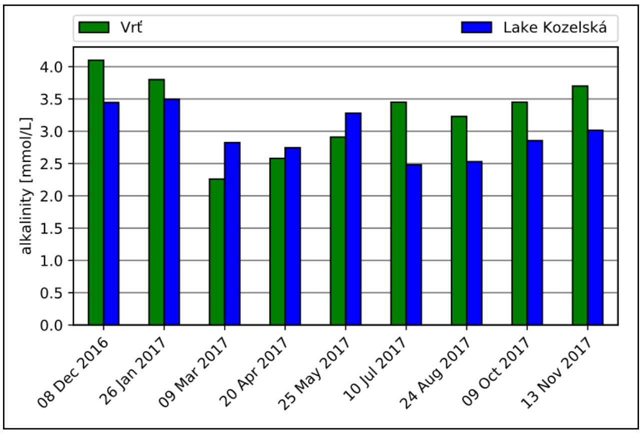

Figure 37: Alkalinity of Lake Vrť and Lake Kozelská during measuring period

Seasonal alkalinity development during the measuring period is shown in 37. High values of alkalinity were proportional to higher concentrations of Ca that creates together with carbonates one of the most important buffering systems in water (Pitter, 2015). This occurred especially during the winter months and corresponded to to pH values. In general, higher values of alkalinity were measured in winter in both lakes.

Alkalinity ranged from 2,26 mmol/L in March to 4,1 mmol/L in December inVrť. In Lake Kozelská, the highest determined alkalinity was 3,5 mmol/L in January and minimum value of 2,48 mmol/L was measured in July.

Conductivity

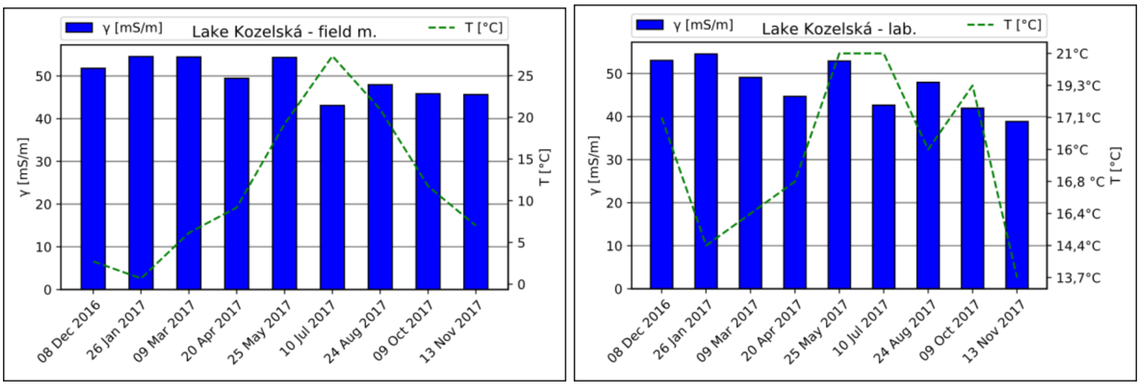

Figure 38: Conductivity of Lake Kozelská measured in the field and in the laboratory

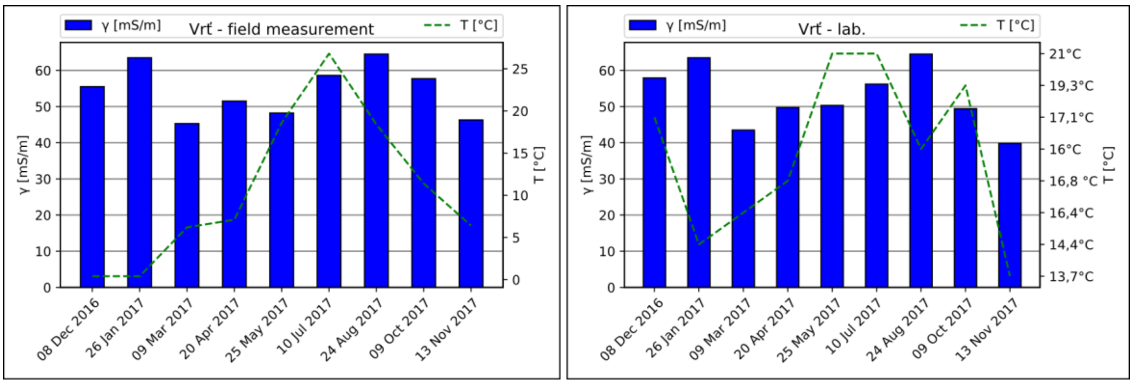

Figure 39: Conductivity of Vrť measured in field and in laboratory

Figure 38–39 represent conductivity measured in the field and in a laboratory. No relation between the data was found. The differencies may result from the use of different devices and conditions during the measurement. The raise of water temperature causes an increase of mobility and number of ions in water. Consequently, the increase of water temperature leads to an increase of its conductivity (Barron and Ashton, 2007). Higher conductivity corresponds with higher concentrations of dissolved charged substances. A failing sewage system could increase the conductivity due to high concentrations of chlorides, phosphates, and nitrates too (Pitter, 2015).

High conductivity was observed for example in January. The results have a positive relation ith higher concentrations of Ca, PO_4, Cl and COD in this lake. Conductivity of Lake Vrť ranged from 45,3 mS/m in July to 64,5 mS/m in August. In Lake Kozelská, the highest value of conductivity reached to 55,5 mS/m in January and March and the minimum value dropped to 43,1 mS/m in July. It may have been caused by low concentrations of Ca and Mg in this month.
An increase of conductivity during winter months might be caused by washing out of e.g. fertilizers, road salts by melting snow or rain or liming of lakes.
All measured values and also all characteristic values did not exceed the limit value for class II of water quality according to ČSN 757 211 (Table 5). Characteristic value was calculated as a mean of 3 the highest values according to the methodology of Capital City Prague. Methods of ČSN 757 221 could not be used in determining the characteristic value due to the low quantity of required values. This research contains only 9 measurements.

Table 5: Limit values of conductivity according to water quality classification (ČSN 757221)

	Water quality classification				
	I	**II**	**III**	**IV**	**V**
conductivity [mS/m]	< 40	< 70	< 110	< 160	≥ 160

7.3.2 Oxygen regime

The concentrations of oxygen dissolved in water are described by Henry's Law. They decrease with the increase of temperature. Therefore, higher amount of O_2 should be found in cold water. Equilibrium oxygen concentrations in water with temperatures are shown in Table 6.

Table 6: Equilibrium oxygen concentration in water depending on temperature at atmospheric pressure 1013,24 hPa (Benson et al., 1980)

T °C	O_2 mg/L	T °C	O_2 mg/L	T °C	O_2 mg/L
0	14,621	11	11,027	22	8,743
1	14,216	12	10,777	23	8,578
2	13,829	13	10,537	24	8,418
3	13,46	14	10,306	25	8,263
4	13,107	15	10,084	26	8,113
5	12,77	16	9,87	27	7,968
6	12,447	17	9,665	28	7,827
7	12,138	18	9,467	29	7,691
8	11,843	19	9,276	30	7,558

9	11,559	20	9,092	31	7,43
10	11,288	21	8,914	32	7,305

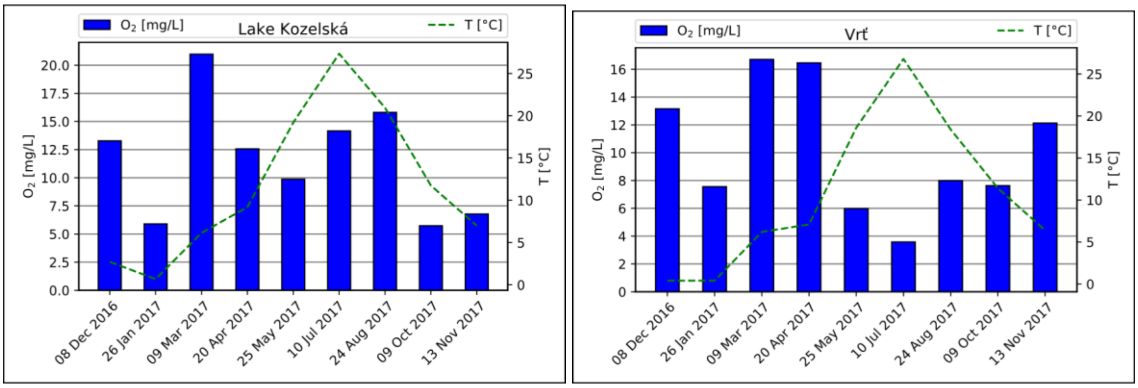

Figure 40: Concentrations of dissolved oxygen Lake Kozelská and Vrť

Both lakes showed relatively similar development of concentrations of dissolved oxygen during the measuring period (Figure 40). Small differences between both lakes were observed in December and April. Lake Kozelská showed higher concentrations during summer months. The reason for it was not the temperature, because it was quite similar in both lakes in summer. Dense population of phytoplankton was found in Lake Kozelská which might have been the cause for the amount of oxygen due to production of O_2 during photosynthesis.

In Vrť, the maximum amount of dissolved reached to 16,7 mg/L in March and April. The lowest concentration of O_2 decreased to 3,58 mg/L under 26,8°C in July. The quantity of O_2 corresponded to all classes of water quality according to ČSN 757211 (Table 7). Class I was determined in 5 months during the monitoring period in Lake Vrť. The values representing class IV of water quality were measured in July. With respect to this parameter, the water quality in Lake Kozelská was even better than Lake Vrť. The results of 6 months during the monitoring period were classified as class I and no values corresponded to class IV or V. Characteristic values in both lakes represent class I. Characteristic value was calculated as a mean of 3 the highest values according to the methodology of Capital City Prague. Methods of ČSN 757 221 could not be used in determining the characteristic value due to the low quantity of required values. This research contains only 9 measurements.

Table 7: Limit concentrations of dissolved oxygen according to water quality classification (ČSN 757221)

Water quality classification					
	I	**II**	**III**	**IV**	**V**
dissolved oxygen [mg/L]	> 7,5	> 6,5	> 5	> 3	≤ 3

7.3.3 Organic substances

<u>COD_{Mn} and BOD_5</u>

Measurements of biochemical oxygen demand (BOD) helps to determine organic water pollution. The parameter indicates bacterial consumption of oxygen used for oxidizing organic compouds in water in a biological way. Bacteria can exhaust oxygen even more than it is replenish from the air, which can be fatal for a number of aquatic organisms - e.g. fish (Hach, 1997).

Chemical oxygen demand (COD) represents than the total concentration of oxygen used for oxidation of organic matter, which can be chemically transformed into inorganic final products using oxidizing agent (potassium permanganate). (Pisarevsky at al., 2005).

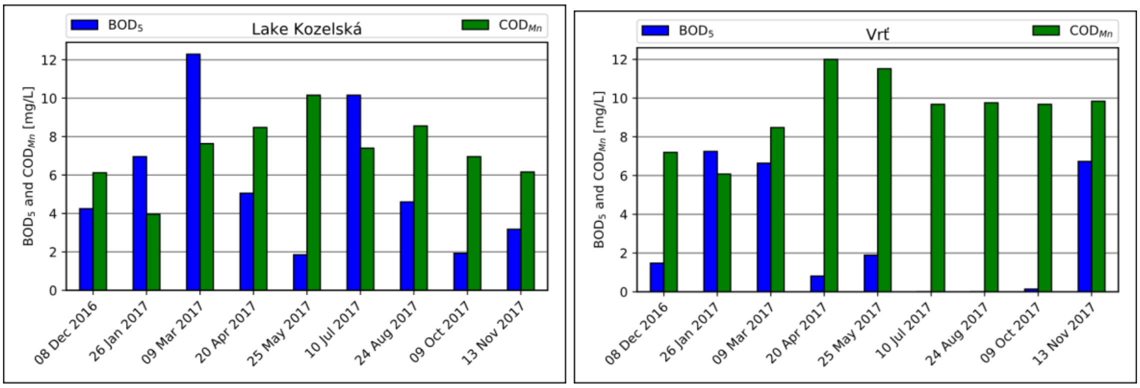

Figure 41: Concentrations of BOD_5 and COD_{Mn} in Lake Kozelská

The results of BOD_5 and COD_{Mn} analyses are depicted in Figure 41. In general, higher values of COD_{Mn} were measured at the beginning of the vegetation period and lasted during the increase of biochemical and decomposition processes under higher temperature in both lakes. Another situation was found in case of BOD_5 measurements.

The amount of COD_{Mn} in Vrť raised up to 10 -12 mg/L at the beginning of the vegetation period. Concentrations of BOD_5 declined to values lower than 2mg/L in both lakes in May and October. The amount of BOD_5 was not determined during laboratory analysis because of the presence of a white precipitate in the samples collected in July and August.

As for Lake Kozelská, the highest concentrations of COD_{Mn} (12 mg/L) were found in April and May, and the minimum values of COD_{Mn} (4 mg/L) were determined in January. However, in January, March and July, the values of BOD_5 exceeded COD_{Mn}, which could be a result of bacterial nitrification as a phenomenon interfering with the correct BOD_5 determination. Due to the principle of determination, BOD_5 values are always lower than COD_{Mn} (Pitter, 1999).

According to ČSN 757221, class I of water quality for BOD_5 concentrations and class II for BOD_5 concentrations were reached in 3 months in Vrť during the measuring period. COD_{Mn} values corresponded to class III during the rest of the year in this lake. In Lake Kozelská, class II of water quality for this parameter was reached for most of the year (Table 4). In Vrť, characteristic values of BOD_5 and COD_{Mn} represented class III. In Lake Kozelská, characteristic value of BOD_5 showed class IV and characteristic value of COD_{Mn} reached class III (Table 8). Characteristic value was calculated as a mean of 3 the highest values according to the methodology of Capital City Prague. Methods of ČSN 757 221 could not be used in determining the characteristic value due to the low quantity of required values. This research contains only 9 measurements.

Table 8: Limit concentrations of COD_{Mn} and BOD_5 according to water quality classification (ČSN 757221)

	Water quality classification				
	I	**II**	**III**	**IV**	**V**
COD_{Mn} [mg/L]	< 6	< 9	< 14	< 20	≥ 20
BOD_5 [mg/L]	< 2	< 4	< 8	< 15	≥ 15

7.3.4 Nutrients

The Nitrogen Cycle

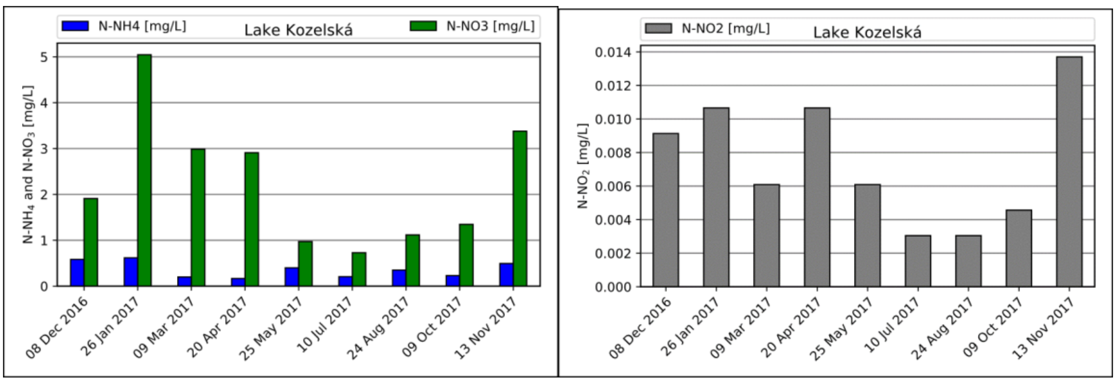

Figure 42: Concentrations of forms of nitrogen in Lake Kozelská

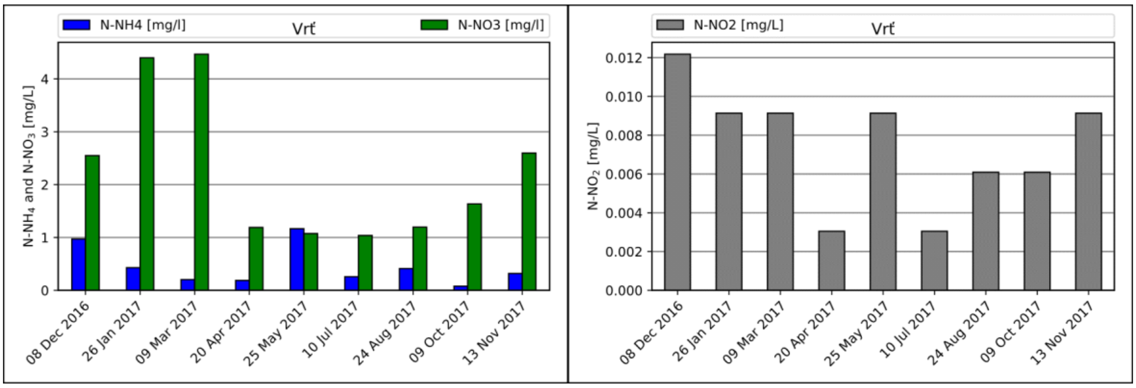

Figure 43: Concentrations of forms of nitrogen in Vrť

In this research, N-NO_3, N-NH_4 and N-NO_2 were determined (Figure 42-43). The amount of N-NO_3 ranged in interval 2 - 4,5 mg/L. N-NO_3 reached the highest concentrations (4,5 mg/L) of all forms of nitrogen, especially at the end of winter and at the beginning of spring. This may have been caused by melting of snow and thus washing down of the soluble substances which could indicate surface contamination. During spring and summer, nitrogen was depleted by plants and heterotrophic microbes. The amount of N-NO_3 was around 2 mg/L through May to August.

Concentrations of N-NH_4 was significantly reduced during whole measured period. The amount of N-NH_4 increased under lower amount of dissolved oxygen in water in May in Lake Vrť (Figure 12). In Lake Kozelská, the concentrations of N-NH_4 were quite constant with slight increase in winter and in May.

The amount of N-NO_2 in water was always very low due to significant instability of this form of nitrogen. Therefore, this form of nitrogen is shown in a different graph. In Vrť, the highest amount of N-NO_2 reached only to 0,012 mg/L in December. In Lake Kozelská, the maximum concentration of N-NO_2 was up to 0.014 mg/l in November.

Concentrations of N-NH_4 did not reach the limit value of class I of water quality according to ČSN757221during March, April, July and October in both lakes. In Vrť, class III for N-NH_4 concentrations was reached in December and May. The amount of N-NO_3 corresponded to class I of water quality for most of the year in both lakes. In general, both lakes represent a hypereutrophic type of a lake.

In Kozelská tůň, characteristic values of N-NH_4 and N-NO_3 reached class II. In Vrť, characteristic values of N-NH_4 represented class III and characteristic values of N-NO_3 showed class II (Table 9). Characteristic value was calculated as a mean of 3 the highest values according to the methodology of Capital City Prague. Methods of ČSN 757 221 could not be used in determining the characteristic value due to the low quantity of required values. This research contains only 9 measurements.

Table 9: Limit concentration of forms of nitrogen according to water quality classification (ČSN 757221)

Water quality classification					
	I	**II**	**III**	**IV**	**V**
N - NH_4 [mg/L]	< 0,3	< 0,7	< 2	< 4	≥ 4
N -NO_3 [mg/L]	< 3	< 6	< 10	< 13	≥ 13

The Phosphorus Cycle

Distribution of P-PO_4 in Lake Vrť and Lake Kozelská shown in Figure 14.

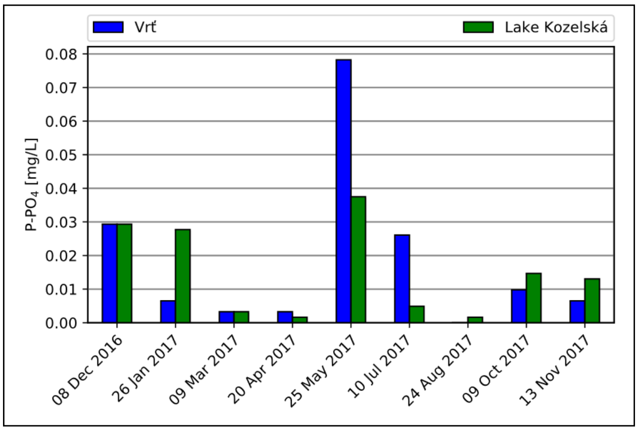

Figure 44: Distribution of P-PO4 in Vrť and Lake Kozelská

P-PO_4 was depleted to 0,003 mg/L due to large productivity of phytoplankton in March, April and August in both lakes. The highest concentrations of P-PO_4 were usually measured during "clear water" periods during low biomass of benthic organisms (Kořínek a kol, 1987), which can be accompanied by low concentrations of dissolved oxygen leading to anoxia allowing phosphorus releases from sediments after reduction of Fe in molecules of $FePO_4$. Higher water temperatures intensify as well biochemical and decomposition processes, which could also cause phosphorus releases into water (Wetzel, 1983).

The highest concentration of P-PO_4 (0,08 mg/L) was observed in Lake Vrť in May. The maximum value in Lake Kozelská (0,04 mg/L) was reached in May as well. Generally, Lake Vrť showed higher amounts of P-PO_4 during May and July. Generally, the development of concentrations of P-PO_4 did not differ much between both lakes during the measuring period.

As both lakes are surrounded by agriculture areas, enhanced values of phosphates could also be a result of fertilizers use in the vicinity of lakes or any other anthropogenic contamination. Concentrations of P-PO_4 decreased probably after depletion of phytoplankton uptake in spring and late summer. The increase of P-PO_4 concentrations was measured again in winter at the end of vegetation period. According to concentrations of P-PO_4 (Table 10), these lakes represented moderately eutrophic types of lakes.

Table 10: Trophic status differentiation in waters according to N a P concentrations (modified by Yang, 2008)

Eutrophic status	TP [mg/L]	TN [mg/L]
Oligotrophic water	0,005 - 0,01	0,25 - 0,6
Moderately eutrophic	0,01 - 0,03	0,5 - 1,1
Eutrophic	0,03 - 0,1	1,0 - 2,0
Hypereutrophic	> 1	> 2

7.3.5 Additional parameters

The determination of concentrations of chlorides, calcium, water hardness, manganese and iron in these oxbow lakes was also a part of this research.

Chlorides Cl^-

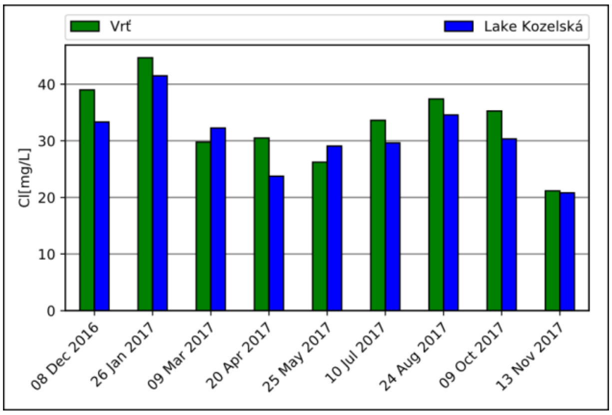

Figure 45: Concentrations of Cl in Vrť and Lake Kozelská

Concentrations of chloride ranged from 20 – 44,5 mg/L in both lakes (Figure 45). In Lake Kozelská, the highest amount of Cl^- reached 41,5 mg/L and in Vrť, it was 44,5 mg/L in January. It could have been related with the salt wash out from roads during snow melting, or fecal pollution coming from nearby villages or chemical industry.

Chlorides are relatively conservative to biochemical processes in water environment (Wetzel, 1983), therefore its increase could be often related to anthropogenic pollution mentioned above. The amount of chlorides decreased during spring and it started to raise again from July.

Minimum concentrations of Cl^- dropped to 20 mg/L in November in both lakes. The measured concentrations did not exceed the limit concentration of class I of water quality according to ČSN 757221 in both lakes during the whole measuring period. (Table 11). Characteristic values of Cl^- reached class I in Lake Vrť and in Lake Kozelská as well. Characteristic value was calculated as a mean of 3 the highest values according to the methodology of Capital City Prague. Methods of ČSN 757 221 could not be used in determining the characteristic value due to the low quantity of required values. This research contains only 9 measurements.

Table 11: Limited concentration of Cl in water quality classification by ČSN 757221

Water quality classification					
	I	**II**	**III**	**IV**	**V**
Cl^- [mg/L]	< 100	< 200	< 300	< 450	≥ 450

Calcium Ca^{2+}

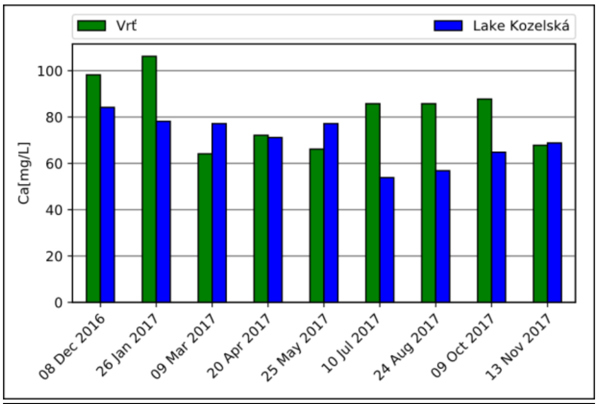

Figure 46:Concentrations of Ca in Vrť and Lake Kozelská

The amount of Ca ranged in the interval 53,8 - 106,2 mg/L (Figure 46). Lake Vrť showed greater changes in concentrations during the year. The maximum amount of Ca increased to 106,2 mg/L in January in Vrť and to 84 mg/L in December in Lake Kozelská. During the winter, the concentrations of Ca showed a positive relation with conductivity and the amounts of Cl and $N\text{-}NO_3$ in both lakes. Higher concentrations of Ca in Lake Vrť could have been also related to anthropogenic sources of pollution. The concentrations did not exceed the limit value of class I of water quality according to ČSN 75 7221 in both lakes during the whole measuring period (Table 12).

Characteristic values reached class I in both lakes. Characteristic value was calculated as a mean of 3 the highest values according to the methodology of Capital City Prague. Methods of ČSN 757 221 could not be used in determining the characteristic value due to the low quantity of required values. This research contains only 9 measurements.

Table 12: Limit concentration of Ca according to water quality classification (ČSN 757221)

	Water quality classification				
	I	**II**	**III**	**IV**	**V**
Ca [mg/L]	< 150	< 200	< 300	< 400	≥ 400

Manganese Mn

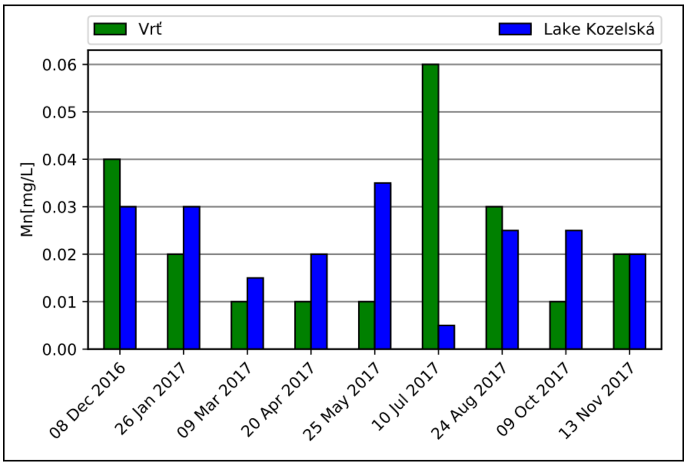

Figure 47:Concentrations of Mn in Vrť and Lake Kozelská

Concentrations of manganese fluctuated during the year (Figure 47). The raise of the amount of Mn was recorded at the end of spring and in summer. It could be caused by the intensification of biochemical processes under higher temperature. Reduced concentrations of oxygen dissolved in water and release of Mn from sediments may also result in the concentrations of Mn in water (Chalupová, 2011). In Lake Kozelská, the highest amount of Mn reached 0,06 mg/L in July and 0,035 mg/L. The lowest concentrations of Mn dropped down to 0,01 mg/L during March and April in Vrť and to 0,02 mg/L in Lake Kozelská. None of the measured values exceeded the limit value of class I of water quality according to ČSN 757221 in both lakes (Table 13). Characteristic values reached class I in both lakes. Characteristic value was calculated as a mean of 3 the highest values according to the methodology of Capital City Prague. Methods of ČSN 757 221 could not be used in determining the characteristic value due to the low quantity of required values. This research contains only 9 measurements.

Table 13: Limit concentration of Mn according to in water quality classification (ČSN 757221)

Water quality classification					
	I	**II**	**III**	**IV**	**V**
Mn [mg/L]	< 0,1	< 0,3	< 0,5	< 0,8	≥ 0,8

Iron Fe

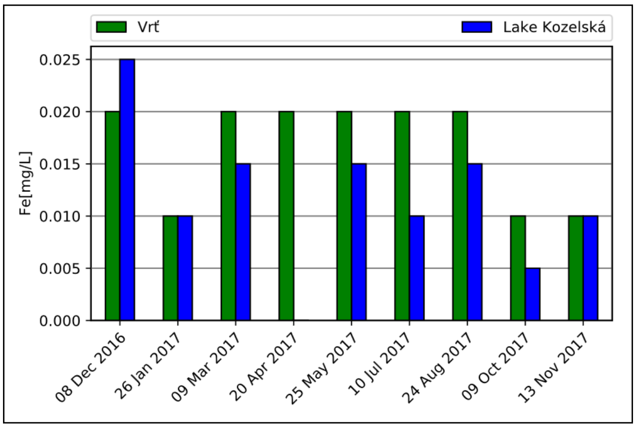

Figure 48:Concentrations of Fe in Vrť and Lake Kozelská

In Lake Kozelská, concentrations of Fe changed a lot during the year (Figure 18). In Lake Vrť, the amount of Fe was stable since March through August and it did never exceed 0,02 mg/L. In Lake Kozelská, the highest concentration of Fe was reached (0,025 mg/l) in December. The increase of concentrations of Fe was observed in spring and summer due to intensification of biochemical processes. The release of Fe from sediment during reduced amount of dissolved oxygen might have caused higher concentrations of Fe in water (Chalupová, 2011).

Limit value of class I of water quality classification according to ČSN 75 7221 was not exceeded in both lakes during the whole measuring period (Table 16). Characteristic values reached class I in both lakes (Table 1614). Characteristic value was calculated as a mean of 3 the highest values according to the methodology of Capital City Prague. Methods of ČSN 757 221 could not be used in determining the characteristic value due to the low quantity of required values. This research contains only 9 measurements.

Table 14: Limit concentrations of Fe according to water quality classification (ČSN 757221)

Water quality classes					
	I	**II**	**III**	**IV**	**V**
Fe [mg/L]	< 0,5	< 1	< 2	< 3	≥ 3

Water hardness

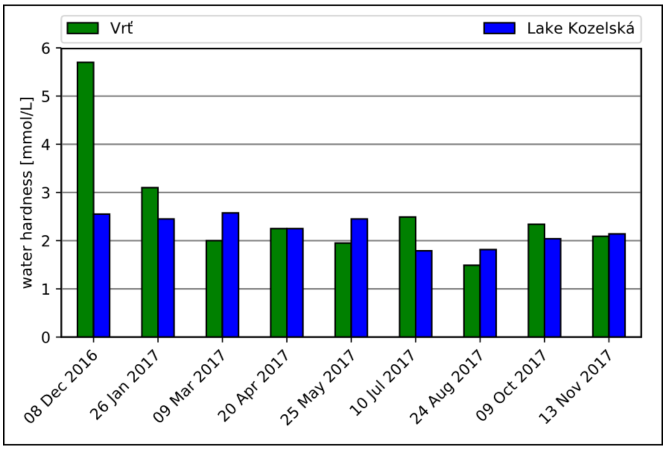

Figure 49: Water hardness of Vrť and Lake Kozelská

Measured values of water hardness were quite stable during the measuring period (Figure 49). The values fluctuated around 2 mmol/L. In Lake Vrť, the exception was found in December at the value of 5,7 mmol/L, which corresponded with higher concentrations of Ca in this month. It is most likely that increased amount of Mg could have been found in that period as well. Water in both lakes was considered a very hard according to classification by U.S.Geological Survey (Table 11).

Table 15: Classification of water hardness modified by U.S.Geological Survey

Classification	hardness in mmol/L
Soft	0 - 0,6
Moderately hard	0,61 - 1,2
Hard	1,21 - 1,8
Very hard	> 1,81

Table 16: Concentrations of measured parameters in Lake Vrť and Lake Kozelská (Characteristic value was calculated as a mean of the 3 worse values according to recommendation of the Czech Hydrometeorological Institute published in the Prague Environment Yearbooks, and according to the Czech National Standard - Classification of Surface Water Quality (757 221)

	Location		Location		Location	
Parameter	**Vrť**	**Lake Kozelská**	**Vrť**	**Lake Kozelská**	**Vrť**	**Lake Kozelská**
	Temperature [°C]		**pH (field)**		**pH (lab.)**	
mean	10,63	11,67	7,27	7,46	7,81	8,06
max	26,80	27,35	9,11	9,12	8,75	8,78
min	0,40	0,69	6,59	6,77	7,31	7,41
stv.dev.	8,98	9,00	0,83	0,82	0,54	0,60
char.value	21,27	22,50	8,17	8,44	8,43	8,72
class						
	conductivity (field) [mS/m]		**conductivity (lab.) [mS/m]**		**BOD_5 [mg/L]**	
mean	54,57	49,68	52,76	47,30	2,77	5,59
max	64,50	54,55	64,50	54,55	7,25	12,30
min	45,30	43,10	39,80	38,85	0,00	1,85
stv.dev.	7,16	4,33	8,45	5,58	3,15	3,61
char.value	62,20	54,45	61,97	53,5	6,87	9,81
class	II	II	II	II	III	IV
	COD_{Mn} [mg/L]		**$N\text{-}NH_4$ [mg/L]**		**$N\text{-}NO_2$ [mg/L]**	
mean	9,36	7,27	0,45	0,36	0,007	0,007
max	12	10,16	1,16	0,62	0,012	0,014
min	6,08	3,96	0,08	0,17	0,003	0,003
std.dev.	1,89	1,78	0,37	0,17	0,003	0,004
char.value	10,37	9,07	0,85	0,56	0,010	0,012
class	III	III	III	II		
	$N\text{-}NO_3$ [mg/L]		**$P\text{-}PO_4$ [mg/L]**		**Cl^- [mg/L]**	
mean	2,24	2,27	0,018	0,015	33,07	30,58
max	4,47	5,05	0,078	0,037	44,67	41,48
min	1,04	0,73	0,000	0,002	21,16	20,81
std.dev.	1,38	1,42	0,025	0,014	7,07	6,02
char.value	3,82	3,80	0,045	0,032	40,35	36,45
class	II	II			I	I
	Ca [mg/L]		**water hardness [mmol/L]**		**Fe [mg/L]**	
mean	81,54	70,23	2,60	2,23	0,02	0,01
max	106,21	84,17	5,70	2,58	0,02	0,03
min	64,13	53,84	1,49	1,79	0,01	0,00
std.dev.	14,91	10,21	1,24	0,30	0,00	0,01
char.value	97,38	79,83	3,76	2,53	0,02	0,02
class	I	I			I	I
	Mn [mg/L]		**alkalinity [mmol/L]**		**O_2 [mg/L]**	
mean	0,02	0,02	3,28	2,96	10,13	11,68
max	0,06	0,04	4,10	3,50	16,71	20,98
min	0,01	0,01	2,26	2,48	3,58	5,74
stv.dev.	0,02	0,01	0,60	0,37	4,67	5,11
char.value	0,04	0,03	3,87	3,41	15,44	16,98
class	I	I			I	I

7.3.6 Statistical evaluation of water quality

Obtained data about water quality of investigated oxbow lakes were analyzed with the Principal component analysis (PCA) using Python 3.6., package sklearn (ter Braak, Šmilauer, 2002). In the case of Lake Kozelská, mean of each parameter from both. Sampling sites was used for analysis in order to evaluate more representative values. Day and month of sampling with initial letter of analyzed lake were used as categorial variable (Herben, Münzbergová, 2003)

At first, values of water quality parameters were standardized due to different units.Resulting ordination diagram (Figure 50) shows the similarity or difference of each measurement in multidimensional space (Chalupová, 2011).

The first axis, which describes the most significant gradient of measured parameters at the dates, explained 35,9 % of variability. The second axis explained 19 % of variability.

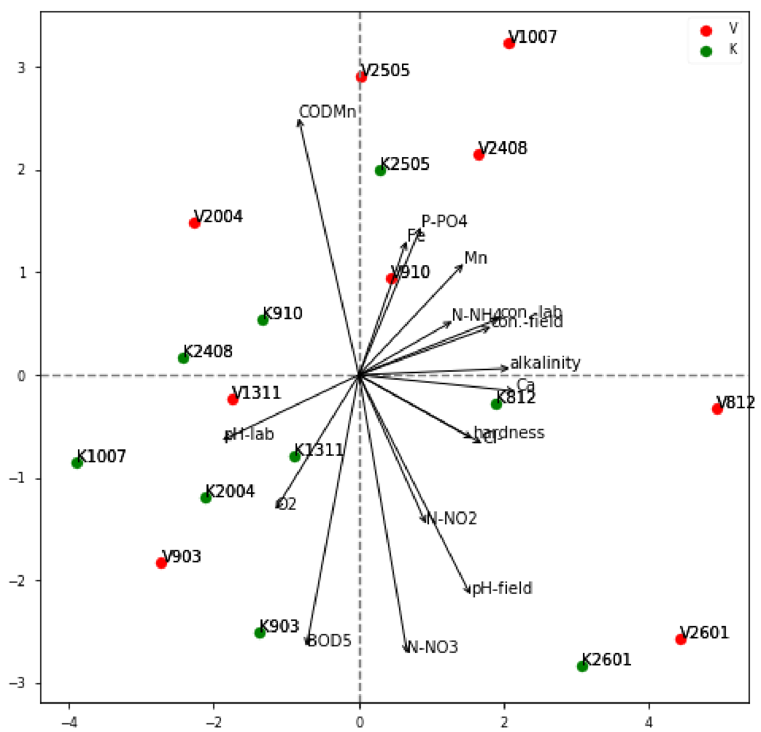

Figure 50: PCA analysis of measured parameters in water

As the length of the vector representing COD_{Mn} in Figure 50 shows, this parameter occurred in the highest relative concentrations. On the contrary, the amount of $N\text{-}NH_4$ was represented by the lowest relative values. The angle between the vectors represents correlation rate among parameters. Fe with $P\text{-}PO_4$, $N\text{-}NO_2$ with pH measured in the field, water hardness with chlorides, and $N\text{-}NH_4$ with conductivity were most positively correlated parameters. Scores in the graph represent location and date of the sampling. Green scores symbolize Lake Kozelská and red points represent Vrť. Numbers show date of a measurement in a format of day and month (K2004 means sampling in Lake Kozelská on 20th April).

Sample scores located near the center of graph are represented by approximately mean values of measured parameters, on the other hand, scores away from the center show extreme values of certain parameters. This distribution was observed mostly in the case of Lake Vrť. As the proximity of the sample scores corresponds to similar values of the measured parameters, it can be stated that the major difference among samples was found in July, on the other hand, quite similar water quality was found in both lakes e.g. in November, January, March and May.

7.3.7 Analysis of phytoplankton

Four different phylums of phytoplankton were observed in both lakes. Distributions of phylums present in Vrť and Lake Kozelská are shown in Figure 52. Five species of phylum *Cyanobacteria* occurred, namely *Microcystis aeruginosa, Planktothrix agardhii, Snowella lacustris, Snowella litoralis* and *Dolichospermum spiroides*. *Snowella sp,* shown in Figure 51, are usually found in slightly eutrophized reservoirs (Komárek, 1992).

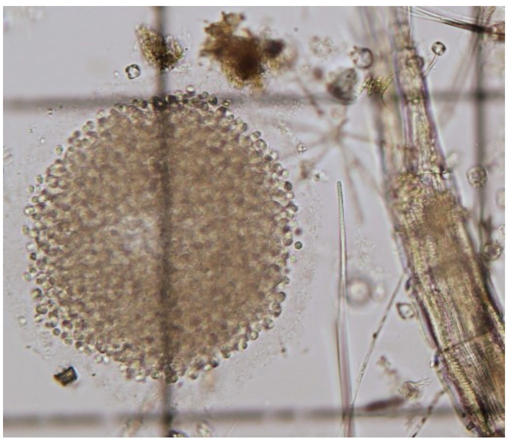

Figure 51:*Snowella* sp, phylum *Cyanobacteria* (20 x magnification, counting chamber of Cyrus I. type, length of a square side was 250 µm)

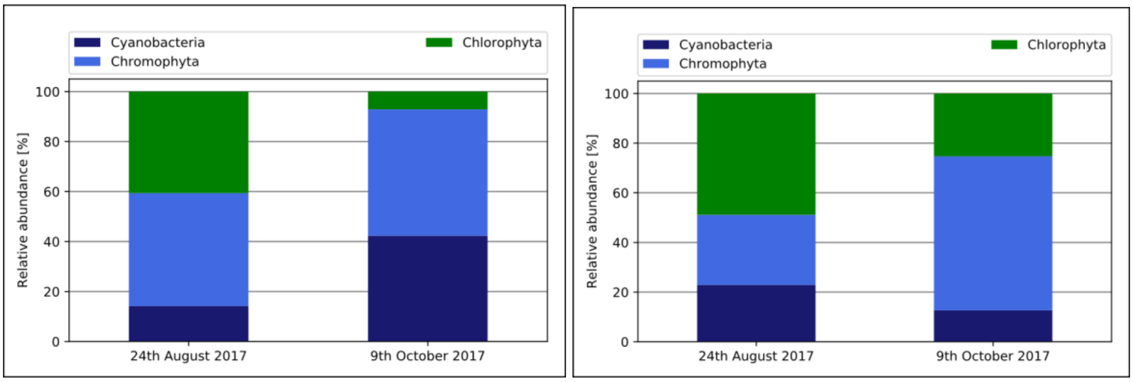

Figure 52: Relative abundance of phytoplankton in Lake Kozelská and Vrť

In August, *Cyanobacteria* represented 15 % of total phytoplankton abundance in Lake Kozelská and this ratio increased to 42 % in October. The growth of *Cyanobacteria* is typical for eutrophic waters (nutrient rich, reduced light), however it may also occur in oligotrophic (nutrient poor, low productivity) systems, although not as frequently as in eutrophic ones (Graham et al., 2008). Three species of class *Bacillariophyceae* representing phylum *Chromophyta* were found (*Synedra acus, Aulacoseira granulate* and *Staruroneis).*

The class *Bacillariophyceae* typically shows a "boom and bust" lifestyle. When conditions in epilimnion (amount of nutrients and light) are suitable, their competitive edge and fast growth enables them to dominate phytoplankton communities. Therefore, they are often classified as opportunistic r-strategists (Furnas, 1990). In Lake Kozelská, high relative proportions of *Chromophyta* were present in August (45 %) as well as in October (51 %). The most numerous phylum was *Chlorophyta*, namely species *Scenedesmus, Desmodesmus, Pediastrum duplex, Pediatrum simplex, Pediastrum boryanum, Phacus elegans, Phacus longicauda, Phacus helicoides, Lepocinclis ovum, Lepocilis texta, Ankistrodesmus gracilis, Chlortetraedron incus, Coelastrum reticulatum, Euglena spirogyra, Trachelomonas volvocinopsis* and *Volvox.*

Chlorophyta is a division of green algae, which can provide nutrients and protect bacteria from environmental stress, such as drying, predation, and damaging radiation. Therefore, they can potentially harbor and enhance the survival of pathogenic bacteria released into the environment. (Byappanahalli, 2003).

In August, in Lake Kozelská, phylum *Chlorophyta* reached 40 % of total phytoplankton abundance. During October, their ratio was much lower, only 7 %.

In Lake Vrť, *Cyanobacteria* showed 23 % in August, and only 13% of total phytoplankton abundance in October. In August, the phylum *Chromophyta* represented

28 %, on the other hand, the relative abundance reached 61 % in October. In August, *Chlorophyta* showed 49 % of abundance, which dropped to 26 % in October.

Phytoplankton abundance in ind./mL and in relative abundance is shown in Table 17. Concerning phylum *Chromophyta*, 62 % of the class *Bacillariophyceae* were found in Lake Vrť in October. It is the highest quantity of all analyzed classes during both months. In Lake Kozelská, the abundance of the class reached high numbers (29 %) too. Similar quantities of class *Chrysophyceae* occurred in this lake in October, whereas in August, this class was not present. In Vrť, *Chrysophyceae* were not observed at all.

Phylum *Dinozoa* was represented by a low abundance (0,3-0,4 %) in Lake Kozelská, and none of the phylum was observed in Vrť during both months. Generally, the density of *Dinozoa* was not significant in comparison with other classes. Therefore, the quantity is demonstrated only in Table 17. Phylum *Dinozoa* usually occurs in lakes with moderate or good water quality (Kalff, 2002).

Figure 53: *Scenedesmus, Desmodesmus*, phylum *Chlorophyta* (20 x magnification, counting chamber of Cyrus I. type, length of a square side was 250 µm)

Table 17: Phytoplankton abundance (composition of species represents only the sample, not whole lake)

Phytoplankton	August 2017				October 2017			
	Vrť [ind./ml]	Vrť [%]	Lake Kozelská [ind./ml]	Lake Kozelská [%]	Vrť [ind./ml]	Vrť [%]	Lake Kozelská [ind./ml]	Lake Kozelská [%]
Phylum Cyanobacteria	1200	22,9	1410	14,1	2200	12,6	7850	42,2
Phylum Chromophyta:								
Class Bacillariophyceae	1480	28,2	4500	45,0	10800	62,1	5380	28,9
Class Chrysophyceae	0	0,0	0	0,0	0	0,0	4000	21,5
Phylum Chlorophyta	2560	48,9	4040	40,4	4400	25,3	1320	7,1
Phylum Dinozoa	0	0,0	40	0,4	0	0,0	53	0,3

7.3.8 Analysis of zooplankton

In water samples from Lake Kozelská, five different phylums of zooplankton occurred in water samples. In Vrť, just four phylums were observed. Distribution of zooplankton is shown in Figure 55.

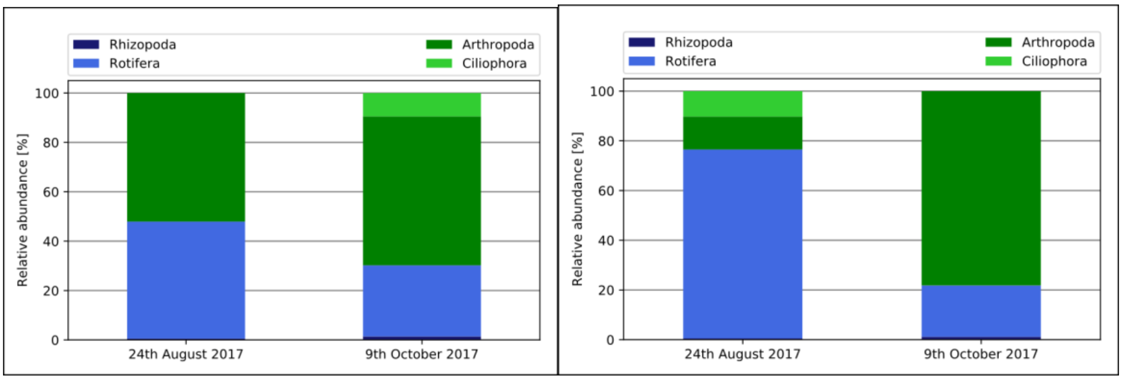

Figure 54: Relative abundance of zooplankton in Lake Kozelská and Vrť

Species distribution in Lake Kozelská varied during both months. The two most abundant phylums of zooplankton occurred in August, namely *Arthropoda* (52 %) and *Rotifera* (47 %). Organisms of the phylum *Rotifera* are r-strategists of small size with short life cycles and tolerance to many environmental factors (Neves, 2003). *Bosmina longirostris* representing phylum *Artrhopoda,* often lives in epilimnion, where is the highest concentration of algae that are consumed by them (Miller, 2000). In this lake, phylum *Rhizopoda* showed only minimal relative abundance of 0,5 % of total zooplankton in August. In October, the most numerous phylum was *Arthropoda* (60 %), *Rotifera* were represented by 29 %, and *Rhizopoda* showed minimal numbers (1,4 %). Compared to the results from August, *Ciliophora* appeared with relative abundance of 9,5 % in October. Parasitic *Ciliophora* may cause morbidity and death of animals (Sacca, 2012).

In Vrť, the most numerous phylum was *Rotifera* with its 76 % in August. The minority phylums *Arthropoceda* and *Ciliophora* showed relative abundance of 10 % and 13 % of total zooplankton quantities in August. *Rhizopoda* represented just negligible portion (0,5 %). Its relative abundance slightly increased to 1,2 % in October. The most common phylum was *Arnthropoda* (78 %) and phylum *Rotifera* that showed 20 % in October.

Generally, phylums *Rotifera* and *Artrhopoda* were the most abundant in both lakes with a permanent presence of fish. .

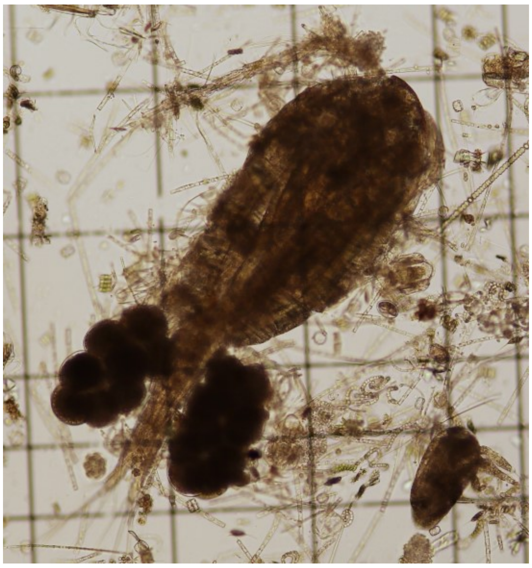

Figure 55: ***Acanthocyclops vernalis*****, phylum** ***Anthropoda*** **(20 x magnification, counting chamber of Cyrus I. type, length of a square sode was 250 µm)**

Species distribution of different phylums of zooplankton in both lakes is shown in Table 18. *Rhizopoda* was the least abundant phylum with the only one present species, *Arcella vulgaris*. Phylum *Ciliophora w*as represented by three species - *Coleps hirtus, Vorticella* and *Codonella cratera. Concenrning* phylum *Rotifera,* nine species were observed, namely *Polyartha vulgaris*, *Brachionus angularis, Brachionus calyciflorus, Keratella cochlearis, Keratella quadrata, Lecane luna, Cephalodella gibba, Asplanchna herricki* and *Kellicottia longispina.* Low quantity of *Aplanchna sp* may enable higher densities of *Brachionus* sp, because of it its predacy.. Species *Lecane luna*, *Cephalodella gibba, Asplanchna herricki* and *Kellicottia longispina* were not observed in Vrť at all. In Lake Kozelská, species mentioned above occurred with a minimal abundance (0,3-2,5 % of zooplankton species). Specie *Daphnia sp* was represented by a very low density of total zooplankton in both lakes during both moths with an exception of one sampling in Vrť, when it reached 9,7 % of all species in October. Eight species of phylum *Anthropoda* was identified, specificaly *Bosmina longirostris, Acanthocyclops vernalis, Ceriodaphnia reticulata, Ceriodaphnia quadrangula, Daphnia longispina, Daphnia cucullata, shells of Ostracoda sp.* and subclass *Copepoda* as well.

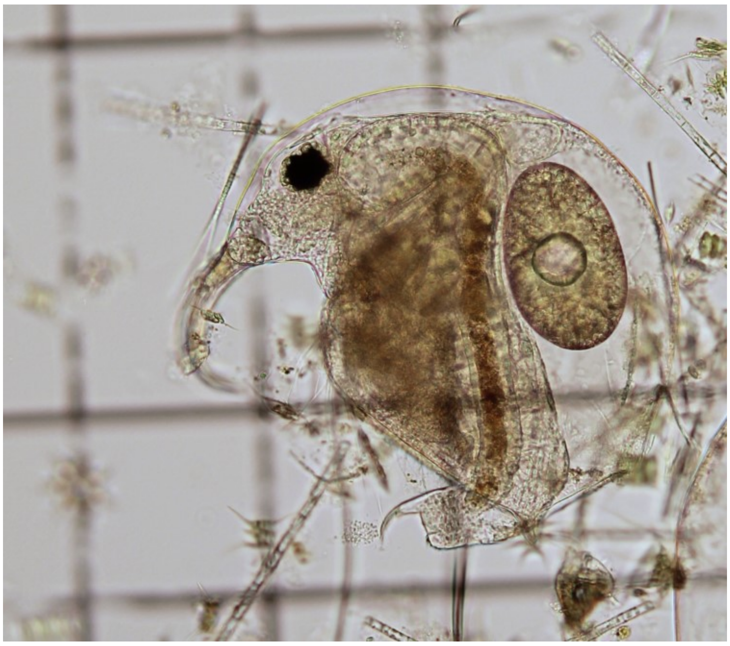

Figure 56: ***Bosmina longirostris*****, phylum** ***Arthropoda*** **(20xmagnification, counting chamber of Cyrus I. type, length of a square side was 250 µm)**

As most of species *Daphnia* consume phytoplankton, they contribute to a better water quality, however, their quantities are reduced by fish. The species is often used as a testing organism in toxicity tests (Forro, 2008). Species *Daphnia longispina* is shown in Figure 59. Other species of the phylum *Artrhopoda* are shown in Figure 57 and 58.

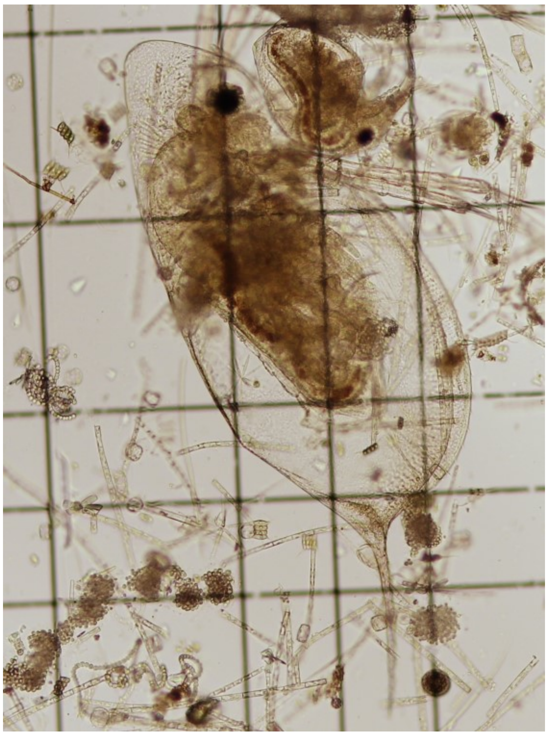

Figure 57: ***Daphnia longispina*****, phylum** ***Arthropoda*** **(20xmagnification, counting chamber of Cyrus I. type, length of a square side was 250 µm)**

Table 18: Zooplankton abundance (composition of species represents only the sample, not whole lake)

Zooplankton	August 2017				October 2017			
	Vrť [ind./ml]	Vrť [%]	Lake Kozelská [ind./ml]	Lake Kozelská [%]	Vrť [ind./ml]	Vrť [%]	Lake Kozelská [ind./ml]	Lake Kozelská [%]
Phylum Ciliophora								
Coleps hirtus (O.F.Müller 1786)	40	9,0	0	0,0	0	0,0	0	0,0
Vorticella Linnaeus, 1767	0	0,0	4	0,7	0	0,0	40	3,9
Codonella cratera (Leidy, 1877)	0	0,0	10	1,7	0	0,0	20	2,0
Phylum Rhizopoda				0,0		0,0		0,0
Class Testacea		0,0		0,0		0,0		0,0
Arcella vulgaris Ehrenberg, 1830	2	0,5	2	0,3	10	0,6	9	0,9
Phylum Rotifera				0,0		0,0		0,0
Polyarthra vulgaris Carlin, 1943	248	55,9	78	13,6	60	3,9	43	4,2
Brachionus angularis Gosse, 1851	20	4,5	12	2,1	20	1,3	11	1,1
Brachionus calyciflorus Pallas, 1766	0	0,0	34	5,9	0	0,0	20	2,0
Keratella cochlearis (Gosse, 1851)	28	6,3	36	6,3	100	6,5	28	2,8
Keratella quadrata (O. F. Müller, 1786)	2	0,5	11	1,9	0	0,0	26	2,6
Lecane luna (Müller, 1776)	0	0,0	2	0,3	0	0,0	6	0,6
Cephalodella gibba (Ehrenberg, 1830)	0	0,0	10	1,7	0	0,0	0	0,0
Asplanchna herricki de Guerne, 1888	0	0,0	0	0,0	0	0,0	25	2,5
Kellicottia longispina (Kellicott, 1879)	0	0,0	0	0,0	0	0,0	24	2,4
Phylum Arthropoda:		0,0		0,0		0,0		0,0
Subphylum Crustacea	52	11,7	187	32,6	680	43,9	383	37,6
Bosmina longirostris (O. F. Müller, 1776)	8	1,8	58	10,1	500	32,3	320	31,4
Acanthocyclops vernalis (Fischer, 1853)	10	2,3	22	3,8	0	0,0	1	0,1
Ceriodaphnia reticulata (Jurine, 1820)	0	0,0	18	3,1	0	0,0	28	2,8
Ceriodaphnia quadrangula (O.F.Müller, 17	0	0,0	4	0,7	0	0,0	9	0,9
Daphnia longispina (O. F. Müller, 1785)	0	0,0	8	1,4	150	9,7	6	0,6
Daphnia cucullate Sars, 1862	0	0,0	8	1,4	0	0,0	9	0,9
Species Ostracoda g.sp (shells) Latreill	0	0,0	1	0,2	0	0,0	0	0,0
Subclass Copepoda	34	7,7	68	11,9	30	1,9	10	1,0

7.4 Sediments quality

7.4.1 Grain analysis

Due to the fact that heavy metals and arsenic are easily bound mainly to fine-grained sediment fractions, a detailed grain analysis for the general assessment of lake contamination was necessary. The grain composition of sediments is also a historical record of processes in the floodplain, such as the extent and the effect of floods or the sediment deposition in the floodplain and oxbow lakes.

The investigated sediment samples were often composed of homogeneous material without significant color changes (Figure 60). The dark color indicated reduction conditions and the presence of organic matter. In most cases, the layers of sediment were classified as fine sand in Vrť and as medium silt in Lake Kozelská.

Figure 58: Homogeneous sediments in core from Vrť

Results of grain analysis of extracted sediment cores are shown in Table 18. The most represented grain size was identified in each sediment layer. Grain size was classified according to the Czech National Standard EN ISO 14668-1 (Table 19). Grain-size distribution curves with marked grain fractions were created as well (Appendix 2).

Table 19: Grain size of sediment samples of both lakes (classification by the Czech National Standard EN ISO 14668-1)

Depth [cm]	Vrť	Lake Kozelská
7	fine sand	medium silt
17	fine sand	medium silt
27	-	medium silt
37	fine sand	fine sand
47	-	fine sand
7	fine sand	-

Table 20: Grain size classification according to the Czech National Standard EN ISO 14668-1

Grain fraction	Size range [mm]
clay	0,002
fine silt	0,0063
medium silt	0,02
coarse silt	0,063

fine sand	0,2
medium sand	0,63
coarse sand	2

Unfortunately, due to too long and too high centrifugation speed,three samples were destroyed (Figure 61).

Figure 59: Broken sample from the centrifuge (photo L.Beranová)

Differences in grain size were found in sediments of Lake Kozelská, especially in the layers of 0 – 27 cm depth, where medium silt was a dominant fraction. In the rest of all investigated layers, more coarse-grained sediment, especially fine sand, was the most represented fraction (Table 18). Fine sand was also the representative grain fraction for each successfully analyzed sediment sample taken from Lake Vrť.

The differences in the grain conditions of the individual localities reflect the natural and anthropogenic processes in the Elbe River. Changes in the deposition of different grain fractions correspond to flood events, when even coarse grain material can be transported into the oxbow lakes. On the contrary, under lower discharge in the river, the drifting ability is smaller, and finer particles may settle down. However, these particles may bind many of pollutants. That is why such material often represents a significant load of heavy metals, which is also supported by a higher organic matter content. The grain composition is also influenced by wash out from surrounding areas, or even wash out of surface sediment layers from the bottom of the oxbow lakes during larger floods when the former meanders may become again a riverbed for the flooding river. A bank stability can play a major role as well.

The surface layers of sediments obtained in Lake Kozelská contained a substantial portion of fine sand material probably originating from the wider canal on the east side connecting the Elbe River and the oxbow lake. This sample was located in the western part of Lake Kozelská. In the eastern part, only very coarse sand was found. This kind of sediment was not collected due to poor ability to bind heavy metals.

Grain-size distribution curves describe the relationship between the diameter of grains and their relative amount in a sample. Figure 62 and 63 show two different grain-size distribution curves. In Lake Kozelská, sediment layer of the depth between 7 and 17 cm contained more medium and coarse silt than the sample from 27-37 cm of sediment depth from Vrť, which corresponded to the results in Table 18. Sediments from Kozelská Lake showed higher portion of fine material, on the other hand, sediment layers from Vrť were characterized especially by coaser fractions. The other curves can be found in Appendix 2.

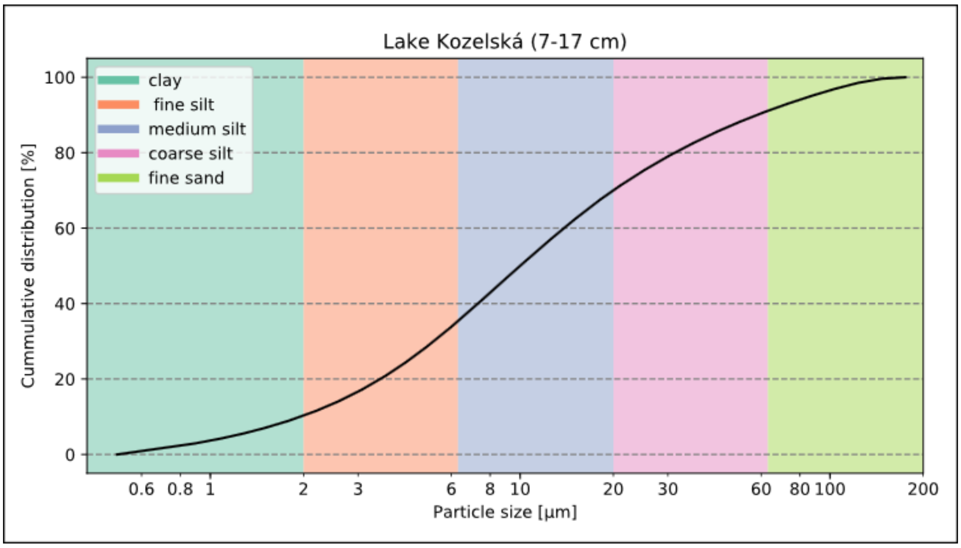

Figure 60: Grain-size distribution curve of the sediment layer 7-17 cm deep from Lake Kozelská

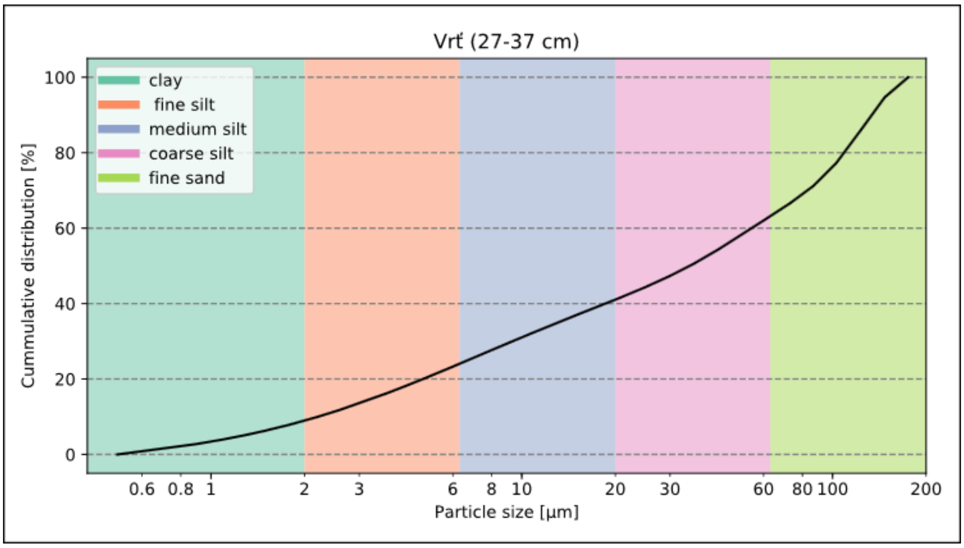

Figure 61: Grain-size distribution curve of the sediment layer27-37 cm deep from Lake Vrť

7.4.2 Chemical analysis

Higher concentrations of metal and arsenic in greater depth of the sediment indicate older contamination of oxbow lakes. Due to low natural background concentrations of the measured metals and arsenic in the non-contaminated subsoil determined in the middle course of the Elbe River (Prange et al., 1997), the enhanced concentrations of selected elements may indicate anthropogenic industrial pollution. The contamination comes from different anthropogenic sources of pollution and the harmful substances can be dissolved or suspended and transported by the river for long distances from the original location (Merian et al., 2004). Extreme hydrological events can also induce remobilization of the deposited material, which can even denude older more contaminated layers of sediment.

<u>Silver (Ag)</u>

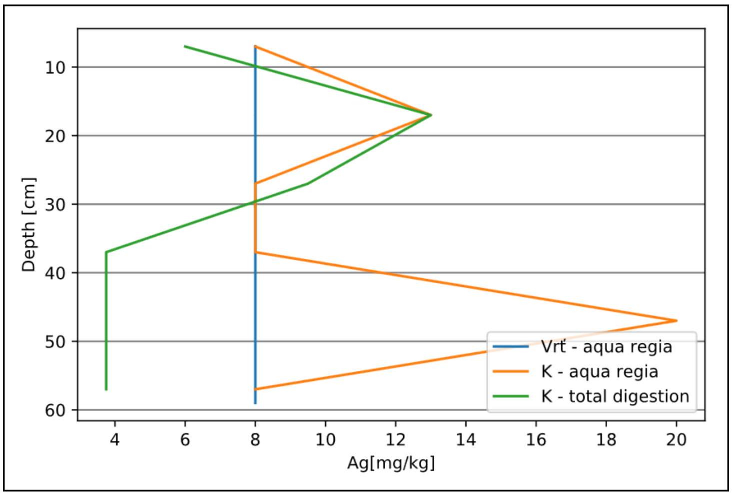

Figure 62: Concentration of Ag in both lakes after using aqua regia leaching and total digestion

Lower concentrations were measured in the whole sediment core in Vrť (Figure 64). The concentrations in each analyzed subsample were lower than the limit of determination, which was 10 mg/kg. In these cases, 75 % of the limit value is usually used for calculations. In the case of silver, this value was 8 mg/kg.

Different situation occurred in Lake Kozelská. Concentrations were much higher, probably due to contamination from Spolana in Neratovice or different anthropogenic source of pollution. In Lake Kozelská, the surface layer showed low amount of Ag. In the depth of 18 cm, this amount increased to 13 mg/kg, but in the depth of 35 cm, it decreased to 8 mg/kg again. These changes could be a result of the different time of contamination of the lake in the past, or dredging the sediment during communist regime, when the excavated layers could be consequently overlapped by younger, less polluted material. In the older sediment layers, the contamination with Ag reached 19 mg/kg.

Arsenic (As)

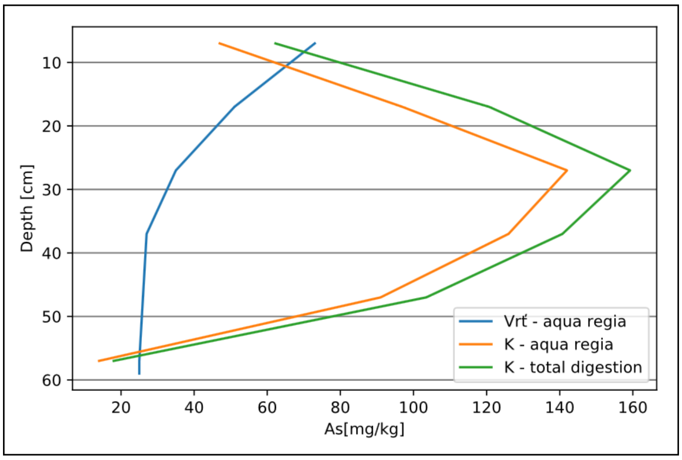

Figure 63: Concentrations of As in both lakes after using aqua regia leaching and total digestion

In Vrť, concentration of As decreased with depth of sediment (Figure 65). The highest content of As was measured in the surface layer (70 mg/kg). Then the values gradually declined to 26 mg/kg in the greatest depth of collected sediment.

In Lake Kozelská, the amount of As was higher and the distribution in the sediment core differed significantly from Vrť. In the surface layer of sediment, the content of As was about 40 mg/kg. The maximum was reached in the depth of 26 cm. Concentrations obtained after aqua regia leaching were a little bit lower than the values measured after total digestion, but the distribution in the sediment core was relatively similar. In the depth of 30 cm in Lake Kozelská, the concentration rapidly decreased.

Aluminium (Al)

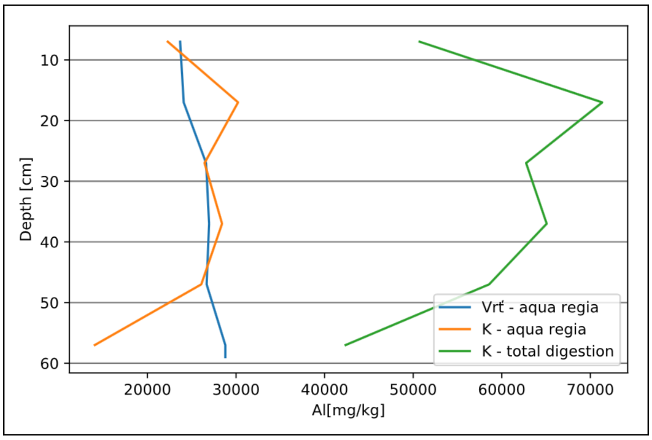

Figure 64: Concentration of Al after aqua regia leaching and total digestion

The distribution of Al contents in sediment core from Vrť was similar as was identified in the case of previously mentioned elements (Figure 66). The values did not change very much in the whole sediment profile, the amount was about 2 700 mg/kg in all subsamples.

A big difference was identified after using aqua regia and total digestion in sediments of Lake Kozelská. The average content of Al after leaching with aqua regia was quite similar to the amount of this element measured in Vrť. On the contrary, the concentrations of Al after using total digestion were much higher. The maximum amount of Al was 7 000 mg/kg in the depth of 17 cm. With grater depth, concentration was decreasing to the same value as in Lake Vrť.

Cadmium Cd

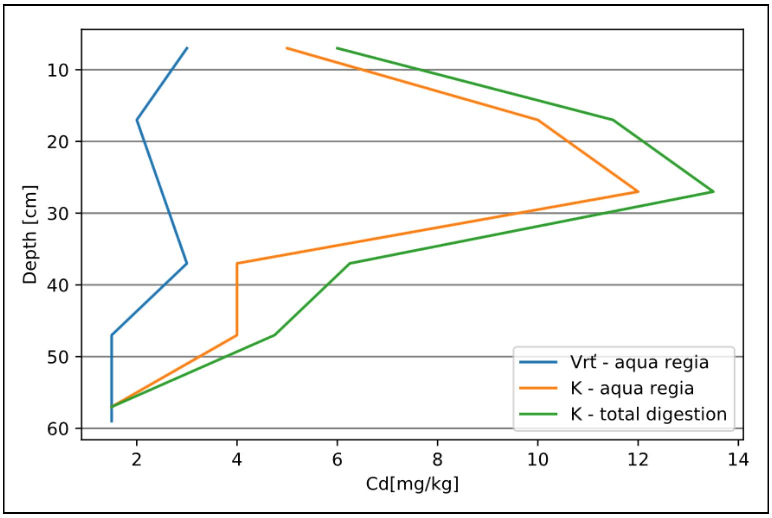

Figure 65: Concentration of Cd after aqua regia leaching and total digestion

In Lake Vrť, the concentrations fluctuated around 2 mg/kg through the whole sediment core (Figure 67). In Lake Kozelská, the contents of Cd were much higher. The amounts after using total digestion reached a little bit higher values, but showed quite the same distribution in the sediment core as after aqua regia leaching. The maximum (13,6 mg/kg) was reached in the depth of 28 cm as in case of Al. Concerning the upper part of the core, the younger layer, the lower concentration of the investigated element. In the youngest surface layer, which was probably deposited in the recent years, the amount of Cd was about 5 mg/kg

Copper (Cu)

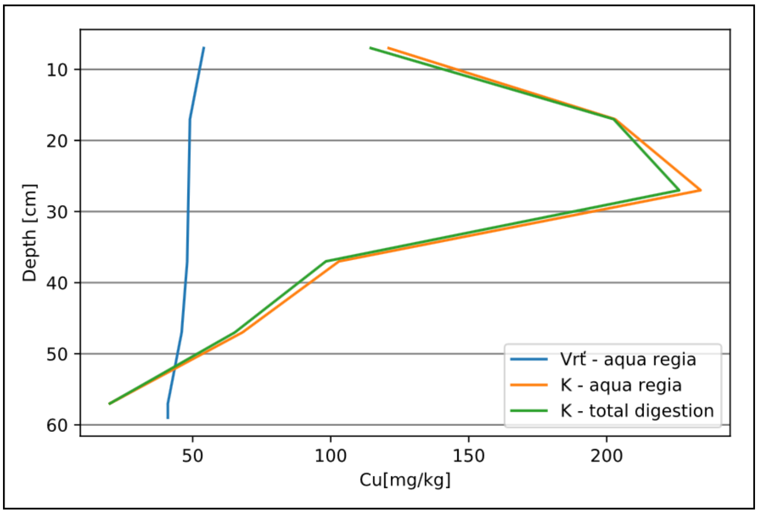

Figure 66: Concentration of Cu in both lakes after aqua regia leaching and total digestion

In Vrť, the content of Cu was relatively constant in each layer (Figure 68). The values fluctuated around 45 mg/kg.

In Lake Kozelská, determined content of copper after leaching with aqua regia and after total digestion reached almost the same values and it has a very similar distribution in the sediment core. In the young surface subsample, the content was only 120 mg/kg and it increased till depth of 26 cm to the concentration of 230 mg/kg. After that, the amount was dropping to the minimal value of 20 mg/kg.

Iron (Fe)

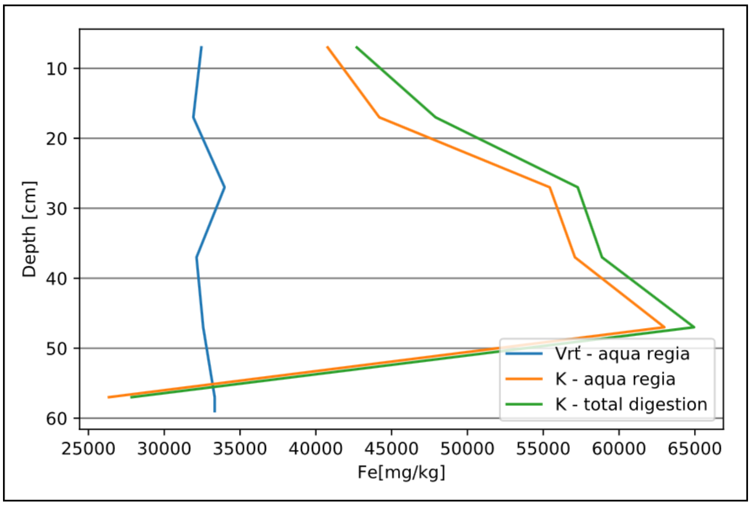

Figure 67: Concentration of Fe in both lakes after aqua regia leaching and total digestion

In Vrť, the amount of Fe present in the sediment fluctuated around 33 000 mg/kg., which was significant especially in the depth from 20 to 40 cm (Figure 69). Above and under this layer, the content of Fe was quite stable (32 000 mg/kg).

In Lake Kozelská, the content of Fe showed different distribution than in the case of previously mentioned elements. A reversal did not occur in the depth of 26 cm as usual, but it was found in the depth of 47 cm, where the maximum concentration of 64 000 mg/kg was found. The lowest concentration was measured in the depth of 0 – 10 cm (40 000 mg/kg), and in the bottom layers (27 000 mg/kg).

Nickel (Ni)

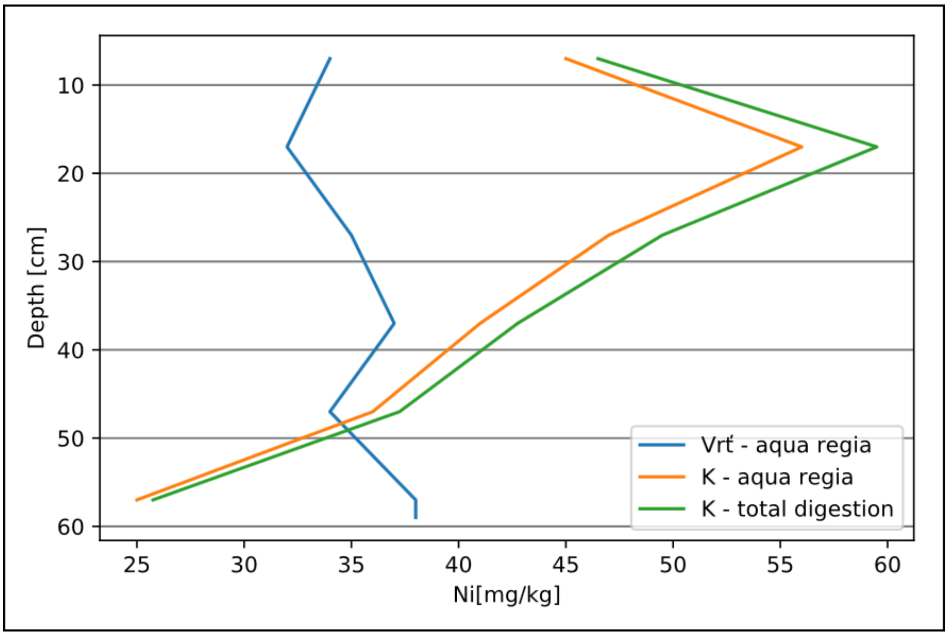

Figure 68: Concentrations of Ni in both lakes after aqua regia leaching and total digestion

In the case of Lake Vrť, the concentration showed different distribution of Ni than was usual in the cases of previously meantioned elements (Figure 70). In general, the amount of Ni slowly increased with the depth. The highest concentration of 38 mg/kg was found in the greatest depth of the sediment core.

In Lake Kozelská, the content of Ni was increasing to the depth of 18 cm where it reached 58 mg/kg. Afterwards, the amount was constantly dropping to the minimum value of 26 mg/kg in the depth of 47-57 cm. The concentration distribution after using both methods of digestion was very similar.

Lead (Pb)

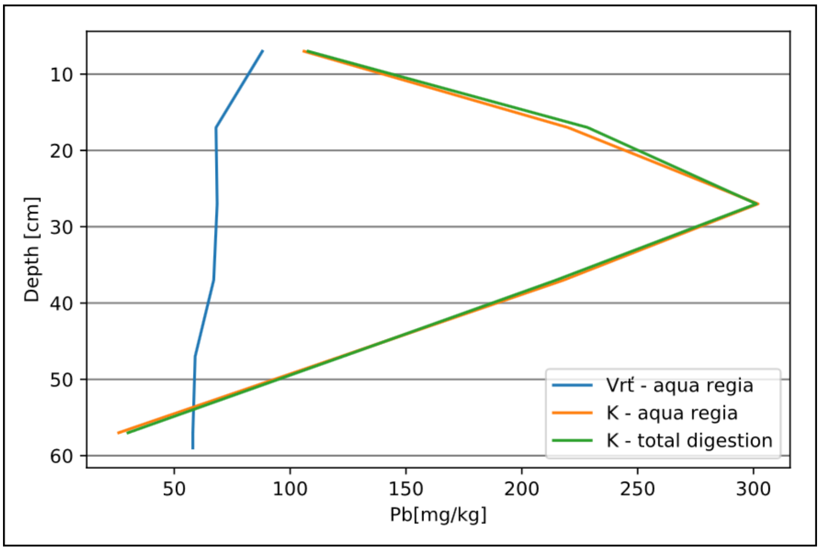

Figure 69: Concentrations of Pb after aqua regia leaching and total digestion

In Lake Vrť, the concentration of Pb was quite stable in whole sediment core (Figure 71). The value oscillated around 60 mg/kg.

On the other hand, in Lake Kozelská, the amount of Pb icreased from the sruface layer to the depth of 28 cm, where the concentration reached 300 mg/kg. Then the amount was dropping to the minimum in the bottom layer.

Titanium (Ti)

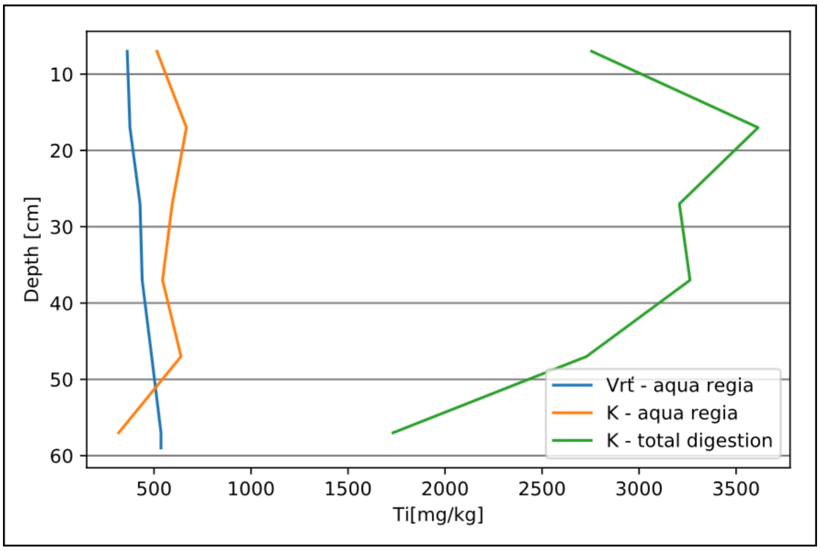

Figure 70: Concentration of Ti in both lakes after aqua regia leaching and total digestion

In Vrť, the concentration of Ti were relatively constant in each sediment layer (Figure 72). The value was about 500 mg/kg.

A big difference was identified after using of both digestion methods in Lake Kozelská. The content of Ti after leaching with aqua regia was quite similar to the concentrations of Ti measured in Vrť. On the contrary, the amount of titanium obtained after total digestion was much higher. The younger layers with the maximum depth of 15 cm showed lower concentrations of Ti. The maximum concentration of Ti reached 3 600 mg/kg in the depth of 18 cm. Downwards, the values decreased but the minimum was still much higher than the lowest content of Ti measured in Vrť.

<u>Zinc (Zn)</u>

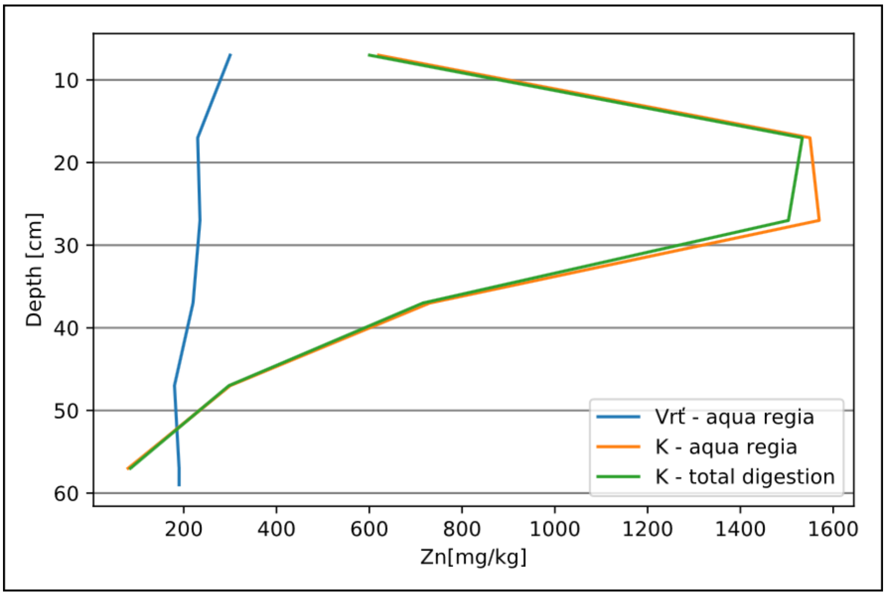

Figure 71: Concentration of Zn in both lakes after aqua regia leaching and total digestion

In the case of Vrť, the amounts of Zn showed similar distribution with low values as was usual in the previously mentioned elements (Figure 73). In Lake Kozelská, the concentration of Zn determined after both digestion methods were almost the same. The content of Zn was increasing to the depth of 18 cm. Then, after reaching the maximum value of 1 550 mg/kg, it gradually dropped to the minimum concentration, which was even lower than the minimum concentration measured in Vrť.

7.4.3 Evaluation of sediments pollution

Sediment pollution was evaluated with Index of Geo-accumulation (I_{geo}) (Müller, 1979) (Forumula 2),

$$I_{geo} = log_2 \frac{C_n}{1,5 * B_n} \tag{2}$$

where C_n is mean measured concentration of determined element (Table 17), B_n is background value of determined element in natural clay material defined by Turekian and Wedepohl (1961). The background values determined by Prange (1997) as more suitable for the Elbe riverbasin were applied as well (Table 22).

This approach enables to classify sediment pollution into six classes of contamination (Tab. 24). Table 21 shows mean values used for calculation of I_{geo} after leaching with aqua regia and after total decomposition as well. In most cases, mean concentrations of elements obtained after total digestion reached the highest values.

Table 21: Mean values of determined elements in sediments

	Mean concentration [mg/kg]										
Location	Core lenght	Ag	Al	As	Cd	Cu	Fe	Ni	Pb	Ti	Zn
Vrť - Aqua Regia leaching	59	8	26517	37	2	47	32816	35	67	452	221
Lake Kozelská - AquaRegia leaching	57	12	24588	86	6	125	47805	42	166	546	808
Lake Kozelská - Total decomposition	57	7,25	58475	100,75	7,3	121	49925	43,54166667	167,5	2883,375	789

Background values of determined elements defined by Turekian and Wedepohl (1961) and by Prange (1997) are shown in Table 22. The biggest difference occurred in the case of silver.

Table 22: Natural background values defined by Prange (1997) and by Turekian and Wedepohl (1961)

	natural background concentrations [mg/kg]											
	Ag	As	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Zn	Al	Ti
GHW - Elbe	0,3	24	0,4	117	32	47600	< 0,3	851	53	29	150	88600	5910
T & W	0,07	13	0,3	90	45	47200	0,4	850	68	20	95	80000	4600

Evaluation according to the Index of Geo-accumulation:

Nearly the half of concentrations of assessed elements did not show any pollution in the investigated sediments. These low values of I_{geo} were calculated for Al, Fe, Ni, Ti after both methods of digestion in Lake Kozelská. In case of Lake Vrť, the sediments were unpolluted with Cu, Pb and Zn.

Generally, I_{geo} calculated from background concentrations defined by Prange (1997) showed lower values, which were classified as less polluted sediments than in the case of using background element contents determined by Turekian and Wedepohl. I_{geo} of elements determined after aqua regia leaching and after total digestion were assessed mostly with the same class of sediment pollution. The methods of digestion did not show as a decisive factor for contamination assessment in the presented research.

The worst results were proven in the case of silver. The contamination with this element was classified as strong or extreme (class 5) using the background concentration determined by Prange (1997), and extreme (class 6) after using the background value defined by Turekian and Wedepohl (1961). In Lake Kozelská, strongly polluted sediment (class 4) was represented by cadmium after both methods of digestion (Table 23).

Table 23: Igeo calculated from aqua regia leaching and total digestion. Diferent background values were used: GHW - Elbe (by Prange) and T & W (by Turekian and Wedepohl)

	I_{geo}					
Element	GHW - **Kozelská** *total digestion*	GHW - **Kozelská** *aqua regia*	GHW - **Vrť** *aqua regia*	T & W -**Kozelská** *total digestion*	T & W- **Kozelská** *aqua regia*	T & W- **Vrť** *aqua regia*
Ag	4,01	4,76	4,15	6,11	6,84	6,25
Al	-1,18	-2,43	-2,33	-1,04	-2,29	-2,18
As	1,48	1,26	0,06	2,37	2,14	0,94
Cd	3,60	3,36	1,97	4,02	3,78	2,39
Cu	1,33	1,38	-0,04	0,84	0,89	-0,53
Fe	-0,52	-0,58	-1,12	-0,50	-0,57	-1,11
Ni	-0,87	-0,93	-1,17	-1,23	-1,29	-1,53
Pb	1,95	1,93	-5,51	2,48	2,47	1,15
Ti	-1,62	-4,02	-4,29	-1,26	-3,66	-3,93
Zn	1,81	1,85	-0,03	2,47	2,50	0,63

Table 24: Classes of sediment pollution according to Igeo (Müller, 1979).

I_{geo} value	I_{geo} class	Sediment pollution
≤ 0	0	unpolluted
≤ 1	1	unpolluted to moderately polluted
≤ 2	2	moderately polluted
≤ 3	3	moderately to strongly polluted
≤ 4	4	strongly polluted
≤ 5	5	strongly to extremely polluted
≥ 5	6	extremely polluted

8 Discussion

8.1 Morphometric characteristics of the lakes

Both investigated lakes corresponded with their morphometric and bathymetric characteristics to the character of oxbow lakes (Havlíková, 2011, Chalupová, 2011, Klouček, 2003, Krýžová, 2007, Turek, 2004; Šobr, 2007). Their area and distribution of depths resulted from erosive and accumulative abilities of the river, even if they were mostly cut from the river artificially during straightening works. Morphometric characteristics of Lake Kozelská, Vrť and other oxbow lakes of the Elbe River are shown in Table 25.

Table 25: Morphometric characteristics of selected oxbow lakes of the Elbe River (source: Chalupová, 2011; Havlíková, 2011)

Parameter	Vrť	Lake Kozelská	Semín	Votoka	Kluk	Němčice	Lžovice	Poděbrady	Václavka	Obříství
lake area (A) [m^2]	25 950	122584	43 360	8 531	18 087	34 099	52 011	18 087	7 323	112 820
perimeter of lake [m]	1 778	3562	1 970	702	1 081	2 062	2 211	1 081	746	4 014
lake volume (V) [m^3]	14 556	125010	30 700	2 814	19 846	21 520	117 987	19 846	3 710	98 000
lengh of lake (L) [m]	821	1509	925	314	516	981	1 040	516	332	1 483
max width [m]	61	100	67	37	46	48	72	46	33	121
mean width [m]	32	81	47	27	35	35	50	35	22	76
max depth Z_{max} [m]	1,6	2,47	1,5	1	2,3	1,3	8	2,3	1,3	1,6
mean depth Z_{ave} [m]	0,56	1,02	0,71	0,33	1,1	0,6	2,3	1,1	0,5	0,9
depth coefficient C	0,35	0,41	0,47	0,33	0,48	0,49	0,3	0,48	0,39	0,54

The area and other morphological and bathymetrical characteristics of Lake Kozelská were most comparable with Lake Obříství (Šnajdr, 2002). The other selected lakes were smaller, narrower and less deep. Therefore, the values of morphometric characteristics of Lake Kozelská were above average among the compared oxbow lakes. The only exception was Lake Lžovice (Chalupová, 2011), which was the deepest of all mentioned lakes. Larger area and greater depths of Lake Kozelská and Obříství could be caused by their location in the lower part of the middle course of the Elbe River, where the river is characterized by its increasing discharge. Concerning Lake Vrť, it was characterized by average values of all its parameters among the oxbow lakes. No measured parameter of Vrť reached extreme values. The morphometric characteristics of Lake Kluk (Havlíková, 2011) and Poděbrady (Chalupová, 2011) were the most similar to the parameters of Vrť.

The differences in the morphometric and bathymetric characteristics of the compared lakes resulted probably from their location within the floodplain, hydrological communication with the river, time of separation, distance from the contemporary riverbed (e.g. siltation of Lake Václavka), discharge in the river (frequency of floods), and human activities in and around the lakes (dredging of sediments, exploratory sand extractions in Lake Lžovice, washout from nearby agricultural areas).

8.2 Water quality

To get a broader context, the researched lakes were compared not only to each other, but also to previous studies available. As in most cases, the development of water quality parameters during the year was not evaluated, average values were used for the comparison (Table 25). Due to a different year and frequency of sampling, the comparison is only indicative.

Most of the **pH** values in Vrť and Lake Kozelská represented neutral or slightly alkaline milieu. Slight increase of pH values was measured at the end of winter and at the beginning of spring, which was driven by the increase of phytoplankton activity, when CO_2 was exhausted from water. In summer, pH in Vrť reached higher values than in Lake Kozelská, which was accompanied by significantly higher concentrations of dissolved oxygen in Vrť during these months.

In Lake Kozelská and especially in Lake Vrť, high values of **conductivity** were observed during winter, especially in January, which corresponded to higher concentrations of Ca, Cl, and $N\text{-}NO_3$. The increase of conductivity during winter months might be caused by wash out from arable land, usage of road salts or liming of lakes (Chalupová, 2011). Higher concentrations of chlorides and phosphates could also indicate wastewater pollution (Pitter, 2015).

Concerning other oxbow lakes, Lake Kozelská showed similar values of conductivity as Lakes Obříství and Němčice (Chalupová, 2011). The highest values of conductivity measured in March corresponded to the periods of increased concentrations of inorganic nitrogen, chlorides, calcium and COD_{Mn}. Higher average values of this parameter were found in Lake Obříství also during the research carried out in 2000 and 2001 (Šnajdr. 2002).

In comparison to Vrť, Lake Kozelská showed higher concentrations of **dissolved oxygen** during summer months, which could be a result of dense phytoplankton population that was found in the lake. In general, lower concentrations of oxygen during warm season, which was significant especially in Lake Vrť, corresponded to higher temperatures, when the solubility of oxygen in water is lower, and to the intensification of decomposition processes when oxygen is consumed. The decrease of oxygen concentrations could also correspond to the "clear water" period after the fall of phytoplankton, which was accompanied by high concentrations of phosphorus in water (Lellák, Kubíček, 1991). The most similar trend of oxygen concentrations as in Vrť was determined in Němčice (Chalupová, 2011), which could be a result of similar processes in the lakes.

Nitrogen supply is a determining factor for the primary production of aquatic ecosystems, where it is consumed by plants and heterotrophic microbes (Kalff, 2002). Generally, atmospheric nitrogen deposition has globally doubled, mainly because of human activities as combustion of fossil fuel, using of fertilizer, increase of human sewage or animal wastes, which can affect nutrient balance in the ecosystems (Jaworski et al. 1997). In Vrť and Lake Kozelská, N-NO_3 indicating non-point sources of pollution reached the highest concentrations from all nitrogen forms especially at the end of winter and at the beginning of spring, which might be caused by wash out of fertilizers during winter precipitation and snow melting before consumption in vegetation period.

In Kluk (Havlíková, 2011) and in Lake Obříství (Chalupová, 2011), the highest average concentrations of **N-NO_3** were observed probably as a result of a local source of pollution. Similarly, the highest concentrations of nitrogen were determined at the end of winter and at the beginning of spring. A similar development of **N-NO_3** concentrations during a year as in Lake Kozelská and Vrť was also found in Němčice. The other compared oxbow lakes showed a different trend.

Concerning **N-NH_4**, Lake Kozelská and Vrť showed extremely high concentrations of this nitrogen form in comparison to other oxbow lakes.

Phosphorus plays a major role in metabolism of aquatic ecosystems, and it could be a limiting factor for the productivity and eutrophication of lakes (Cullum et al., 2006). From this point of view, the most important is orthophosphate phosphorus (PO_4), as it can be consumed by autotrophic organisms. However, more than 90 % of phosphorus in water exists as organic phosphates and cellular components in biota, dead organisms or adsorbed in inorganic compounds deposited in sediments (Wetzel, 1983). In both lakes, P-PO_4 was depleted due to large productivity of phytoplankton in March, April and August. The highest concentrations of P-PO_4 were measured in May probably during “clear water” period after the fall of planktonic organisms (Kořínek a kol, 1987), when low concentrations of dissolved oxygen caused bottom anoxia allowing phosphorus releases from sediments e.g. after reduction of Fe in molecules of $FePO_4$.

Higher water temperatures also intensify biochemical and decomposition processes, which could also cause phosphorus releases into water (Wetzel, 1983). The increase of $P\text{-}PO_4$ concentrations was measured also in winter at the end of vegetation period. As both lakes were surrounded by agriculture areas, the enhanced values of phosphates could also be a result of fertilizers use in the vicinity of the lakes or other anthropogenic contamination.

Lake Obříství showed the most similar concentrations of **$P\text{-}PO_4$** as were stated in Lake Kozelská and Vrť. As the lake was surrounded by agricultural fields, a pasture and a village, its water was probably heavily influenced by these activities around the lake.

In both lakes, the highest concentrations of **chlorides** were reached in January, which could be related to salt wash out from roads during snow melting, or it could indicate fecal pollution coming probably from nearby villages or chemical industry, as chlorides are relatively conservative to biochemical processes in aquatic environment (Wetzel, 1983). The amount of chlorides decreased during spring and it started to raise again from July.

Concentrations of **calcium** usually decrease due to precipitation of $CaCO_3$ during summer months as a result of intense photosynthesis of planktonic organisms causing major epilimnetic decalcification after CO_2 depletion (Wetzel, 1983). Lake Vrť showed greater changes in concentrations during the year. The maximum content of Ca was determined in Vrť in January, and in Lake Kozelská in December. During winter, the concentrations of Ca showed a positive relation with conductivity and the contents of Cl^- and $N\text{-}NO_3$ in both lakes. Higher concentrations of Ca in Lake Vrť could also be related to an anthropogenic source of pollution.

Manganese is an important element for many processes in living organisms, for that reason, its distribution is influenced by biochemical processes in aquatic environments. Mn is excluded into water by animals and plants, it is also released during decomposition of dead bodies or from sediments under the lack of oxygen in water (Pitter, 2015), which was probably determined in Vrť in July. The raise of concentrations was recorded at the end of spring and in summer due to intensification of biochemical processes under higher temperature in both lakes. The increase of concentrations of **iron** was observed in spring and summer due to intensification of biochemical processes in both lakes. The release of Fe from sediment during reduced amount of dissolved oxygen might cause higher concentrations of Fe in water (Chalupová, 2011).

Table 26: Average values of selected parameters of water quality in different oxbow of the Elbe River (Havlíková, 2011; Chalupová, 2011, Chalupvová, 2003; Klouček, 2002; Krýžová, 2007).

Parameter	Lake Kozelská 2017	Vrť 2017	Vrť 2004 - 2007	Němčice 2006 - 2007	Lžovice 2006 - 2007	Poděbrady 2006 - 2007	Václavka 2006 - 2007
O_2	11,68	10,13	**16,60**	8,27	9,86	8,67	9,66
BOD_5	5,59	2,77	4,70	4,50	3,70	3,50	5,20
COD_{Mn}	7,27	9,36	8,20	10,11	5,61	5,40	6,69
$N\text{-}NO_3$	2,27	2,24	2,58	2,10	2,10	2,60	0,10
$N\text{-}NH_4$	**0,36**	**0,45**	0,15	0,18	0,08	0,09	0,05
$P\text{-}PO_4$	0,015	0,018	0,021	**0,540**	0,080	0,040	0,040
conductivity	49,68	52,76	45,00	81,30	46,30	46,50	53,80

Parameter	Obříství 2006 - 2007	Semín 2004 - 2007	Votoka 2004 - 2007	Doleháj 2003	Labiště p.O. 2002	Libiš 2007
O_2	11,46	12,30	12,00	11,47	5,48	7,30
BOD_5	6,30	5,80	5,20	**9,23**	**17,73**	5,80
COD_{Mn}	7,98	9,30	8,30	**25,57**	**20,12**	**18,10**
$N\text{-}NO_3$	**3,10**	0,50	2,67	2,60	0,87	1,50
$N\text{-}NH_4$	0,11	0,08	0,07	1,20	0,59	0,40
$P\text{-}PO_4$	0,070	0,003	0,004	0,020	**0,410**	0,110
conductivity	**69,40**	44,80	**78,10**	49,50	39,50	**129,00**

Concerning the organic pollution of investigated lakes, the water samples from Vrť showed higher organic load than it was determined in Lake Kozelská. In Vrť, the proportion of biodegradable substances was lower, which could be a result of hydrological communication with the Elbe River, or local sources of pollution around Lake Kozelská (fecal pollution, fertilizers). In both lakes, higher values of COD_{Mn} were determined in vegetation period during the rise of biochemical and decomposition processes at higher temperature. However, in Vrť, only minimum values of BOD_5 were measured during these months. The highest values of COD_{Mn} were measured especially in April and May, when these two lakes showed concentrations of 12 mg/L. The most different trend of **COD_{Mn}** concentrations was found in Lake Poděbrady (Chalupová, 2011).

Similar average COD_{Mn} values as in Vrť were determined also in Semín (Havlíková, 2011). In general, Lakes Doleháj (Chalupová, 2002), Labiště pod

Opočinkem (Klouček, 2002) and Libiš (Turek, 2004) were characterized by extremely high concentrations of COD_{Mn} and BOD_5, which could probably result from a local pollution - agricultural fields with a possible application of organic fertilizers as in the cases of Doleháj and Labiště pod Opočínkem, or anthropogenic pollution from chemical industry as in the case of Libiš that is situated nearby Spolana Neratovice chemical plant.

Higher organic pollution was determined also in Lake Němčice, which could be affected by substances produced in paper mill in Hostinné in the past (Chalupová, 2011), or organic pollution from a nearby village lacking any sewage treatment plant, or gardening colony in the vicinity of the lake, as well as autochtonous organic compounds produced by the biota of the lake, which was extremely eutrophicated. During summer, the surface of Lake Němčice was covered with a thick layer of Lemma (Chalupová, 2011).

Hydrological profile Obříství

Obříství is located a few kilometers downstream from Spolana Neratovice chemical plant, which was the biggest source of pollution in the past.

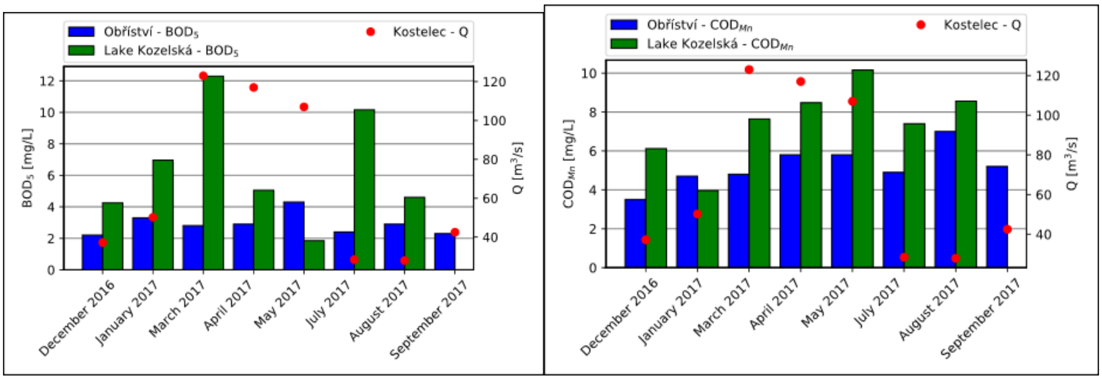

Figure 72: Concentration of COD_{Mn} and BOD_5 in Lake Kozelská and Obříství

In Lake Kozelská, the concentration of BOD_5 and COD_{Mn} were mostly higher than in the Elbe's River hydrological profile Obříství (Figure 74). In March, the biggest difference between BOD_5 concentrations in Lake Kozelská and Obřsítví was found despite relative high BOD_5 values measured in the river. This might be caused by higher microorganism activity in the lake, where are more convenient conditions than in running water, or organic pollution of the lake.

Regarding content of COD_{Mn}, the differences between the concentrations in the river and the lake were smaller. In the lake, the concentrations of COD_{Mn} were mostly higher than in the river. The only exception was determined in January.

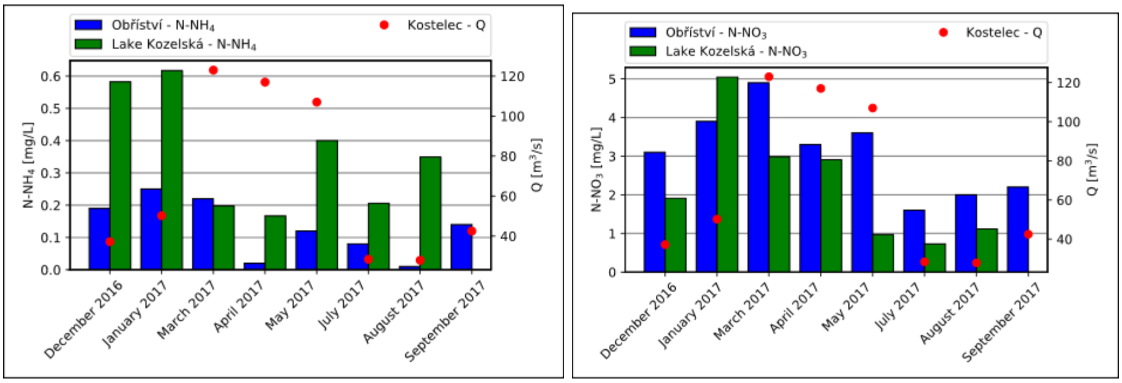

Figure 73: Concentration of N-NH4 and N-NO3 in Lake Kozelská and Obříství

In Lake Kozelská, much higher concentrations of $N\text{-}NH_4$ were observed than in the hydrological profile Obříství (Figure 75). In the river, the lowest content of $N\text{-}NH_4$ was measured at increased discharge when higher oxidation of $N\text{-}NH_4$ could occur at higher flow rates, or the concentrations were diluted.

Regarding concentrations of $N\text{-}NO_3$, the situation was opposite. In the river, $N\text{-}NO_3$ content was mostly higher than in the lake. The concentrations dropped especially in spring and summer, when nitrates were depleted by autotrophic organisms, which was particularly significant in standing water.

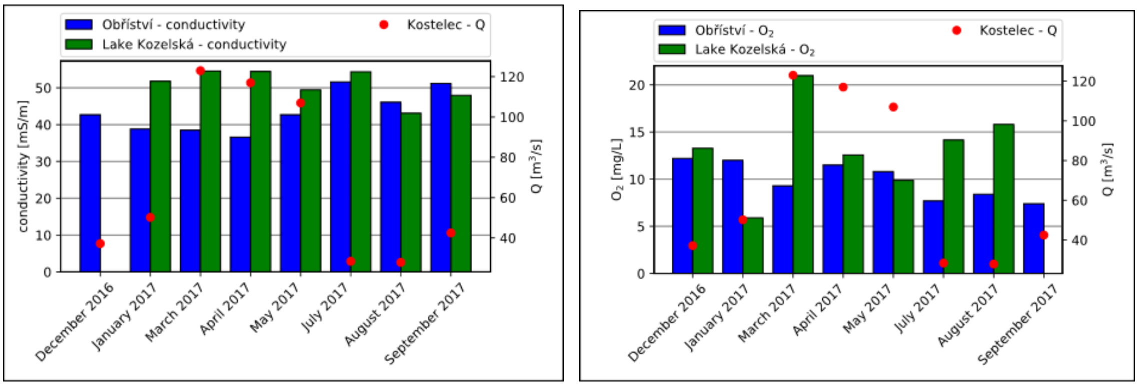

Figure 74: Values of conductivity and concentration of oxygen dissolved in Lake Kozelská and Obříství

Differences in values of conductivity measured in the lake and in the river were quite small (Figure 76). However, in Lake Kozelská, the values were mostly higher than in Obříství due to higher concentrations of Ca or Mn in the lake, especially in spring.

In the lake, dissolved oxygen concentrations increased mainly in March, July and August, which resulted from higher abundance of phytoplankton in these months.

Hydrological profile Lysá nad Labem

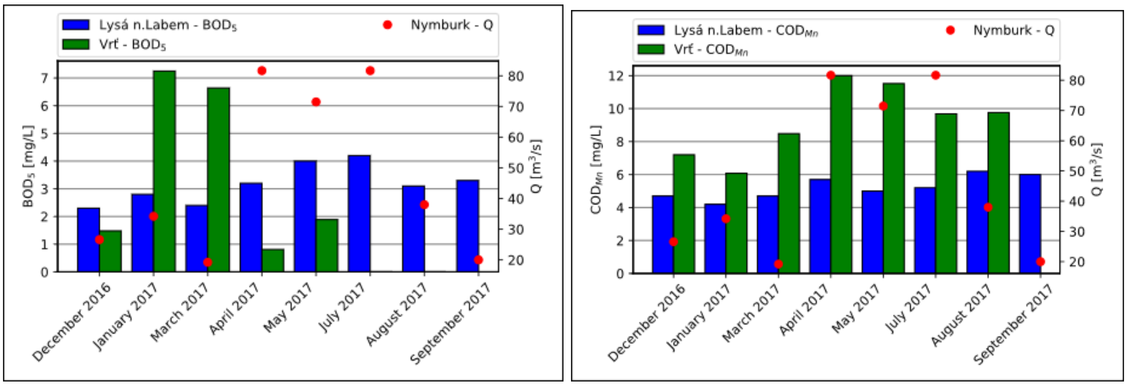

Figure 75: Concentration of BOD_5 and COD_{Mn} in Vrť and Lysá nad Labem

The differences in concentrations of BOD_5 and COD_{Mn} measured in Vrť and the Elbe's River hydrological profile Lysá nad Labem were quite significant (Figure 77). In January and March, the increased BOD_5 values in Vrť were measured probably due to an anthropogenic contamination of the lake. On the contrary, from the end of spring, the BOD_5 was higher in the river, as the decrease of BOD_5 values in Vrť was determined in folowing months, which coudl be a consequence of decomposition of biodegradable substances due to high microorganism activity in the lake. However, the content of organic matter in standing water remained higher during the whole period.

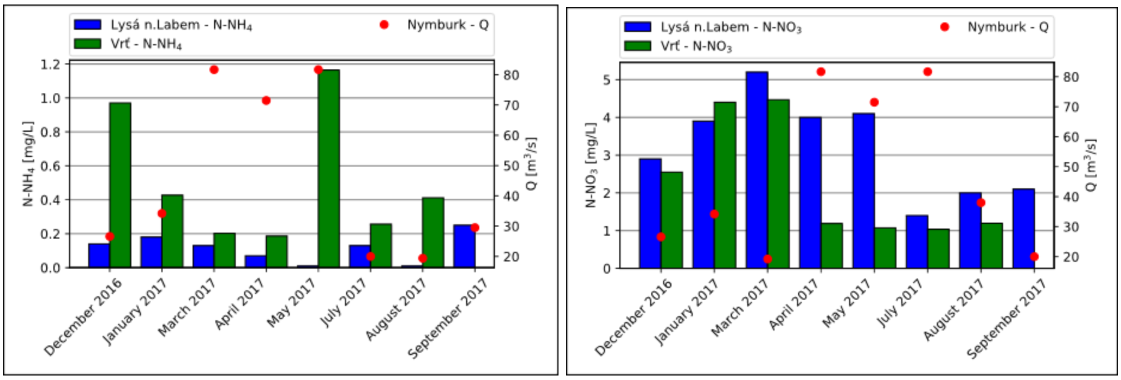

Figure 76: Concentration of $N\text{-}NH_4$ and $N\text{-}NO_3$ in Vrť and Lysá nad Labem

Concerning $N\text{-}NH_4$ concentrations, significant differences between the values measured in the lake and in the river were determined. The biggest differences were measured in December and May, when the highest content of $N\text{-}NH_4$ was recorded in the lake. It was probably not a result of low oxygen saturation, but contamination of the lake.

In the river, the content of N-NO_3 was mostly higher than in the lake as it was found in Obříství too. In the river, the phytoplankton activity was not so intense as in the lakes, therefore N-NO_3 was not depleted.

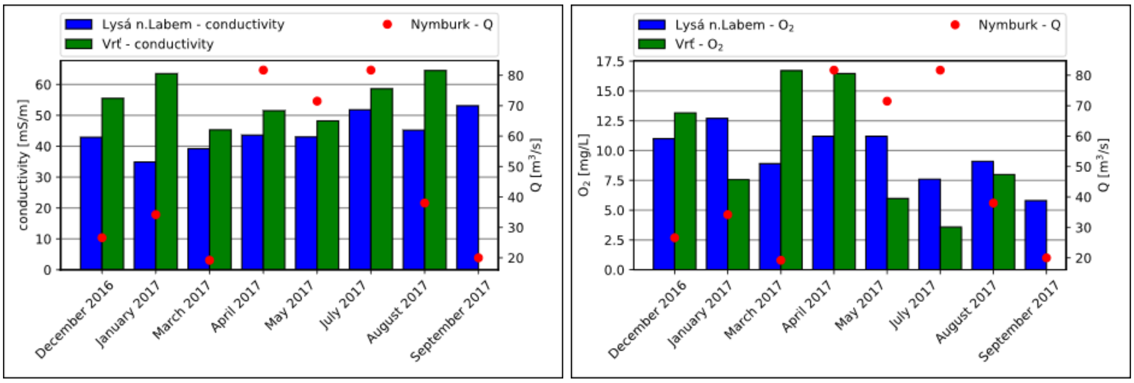

Figure 77: Values of conductivity and oxygen dissolved concentration in Vrť nad Lysá nad Labem

In Vrť, the values of conductivity were allways higher than the values measured in the river (Figure 79). The major differences were found especially in winter. The content of dissolved oxygen fluctuated during the year. In Vrť, concentrations were higher especially in March and April as a result of high phytoplankton activity in the lake.

In general, significant higher concentrations of N-NO_3, which was similar to the results of previous studies of the Elbe River oxbow lakes, were determined in the Elbe River (Chalupová, 2011). On the other hand, the lakes showed usually higher organic pollution, conductivity and oxygen saturation during spring, which was a typical situation determined in eutrophic standing waters.

Although significant differences in the distribution and values of some parameters were observed, the investigated lakes generally correspond to the characteristics of oxbow lakes. Differences were caused especially due to the intensity of the hydrological connection with the Elbe River, the availability of nutrients in the lake and the local contamination of area.

Analysis of zooplankton

The number of zooplankton species determined in the samples taken in the investigated lakes was lower compared to the results of other studies, as the sampling was carried out in pelagic zone, which due to its lover diversity, and usually also availability of nutrients, does not provide habitat to so many species (Ošmera, 1973). The samples collected from the pelagic zone could also be underestimated due to sampling during the day when certain species could hide in the littoral (Hrbáček, 2000a).

Lake Kozelská was represented mostly by *Arthropoda*, which were the most numerous phylum in August and in October as well. Regarding species distribution in Lake Kozelská, *Polyartha vulgaris, Bosmina longirostris* and subphylum *Copepoda* occurred the most. *Daphnia s.p.* was also found in Lake Kozelská. Due to the zooplankton composition, planktivorous fish were probably less represented, however, zooplankton composition could also depend on the age of fish (Hrbáček, 2000a).

The phylum *Arthropoda* dominated in the oxbow lake Kluk as well (Havlíková, 2011), where numerous species of this phylum - *Bosmina longirostris, Daphnia cucullata, Ceriodaphnia pulchella* or *Diaphanosoma brachyurum* were determined. Due to the dominance of *Cyclopida* and *Gymnoplea*, there was a very small proportion of *Rotifera* that could be consumed by older *Copepoda* (Brandl, 2005).

In Vrť, phylum *Rotifera* showed the highest relative abundance in August, while *Arthropoda* dominated in October. Species as *Coleps hirtus*, *Polyartha vulgaris* and subphylum *Copepoda* were determined the most. In the research carried out by Havlíková (2011), *Rotifera* were also dominant in Lakes Semín, Votoka and Vrť during the years 2004 and 2007. In these lakes, large forms of *Cladocera*, e.g. *Daphnia s.p. filtering* effectively phytoplankton, were abundant, which could cause depletion of phytoplankton. Lakes Semín, Votoka and Vrť (Havlíková, 2011) were also characterized by their small area that could influences the zooplankton species composition too. One of the decisive factors was also land use of nearby fields (Havlíková, 2011).

In Lakes Obříství and Doleháj (Chalupová, 2003), the evaluation of zooplankton size structure showed the absence of large species which were eliminated by fish. In Labiště pod Opočínkem (Klouček, 2002), the analysis showed missing large species of zooplankton as a result of fish predation. The size and structure of zooplankton varied also according to the season (Klouček, 2002).

In general, all compared lakes represented mostly eutrophic aquatic ecosystems with zooplankton structure depended on the availability of nutrients (phytoplankton biomass), the character of the water body (shading), as well as the predation pressure of the fish.

8.3 *Sediments quality*

<u>*Comparison of heavy metals and As content in Lake Kozelská and Vrť*</u>

Sediment cores from both lakes showed a different distribution of heavy metals and As in sediment layers (Table 27). In Vrť, the concentrations of almost all investigated elements were lower and showed only small differences of their content with the depth of the sediment core. In Lake Kozelská, the opposite situation was found. Concentrations of determined elements reached mostly higher values and their content changed a lot with the depth of the sediment core. Although these changes could be influenced by dredging of sediments during communist regime or extreme flood in the locality, the high concentrations resulted probably from the situation of the lake about 2 km upstream from Spolana Neratovice, the chemical plant, which represented one of the biggest sources of pollution of the Elbe River in the past. During extreme flooding, suspended matter with adsorbed pollution could be transported even upstream, and then settled in Lake Kozelská, which could probably happen during the flood in 2002.

Regarding different digestion methods, higher concentrations were found after total digestion than after aqua regia leaching, because even very durable forms (silicates) are decomposed with this method including natural background.

Concerning contamination assessment, Indices of Geoaccumulation counted with the use of background values determined by Turekian and Wedepohl (1961) shower higher class of sediment pollution than after using the background values determined in the Elbe riverbasin by Prange (1997), which took into account the specifics of the natural background of the region.

Table 27: : Concentration of determined investigated elements in Lake Kozelská and Vrť after aqua regia leaching

	Vrť [mg/L]										
Depth [cm]	**Ag**	**Al**	**As**	**Cd**	**Cu**	**Fe**	**Ni**	**Pb**	**Ti**	**Zn**	**Hg**
7	8	23688	73	3,0	54	32450	34	88	362	300	0,64
17	8	24091	51	2,0	49	31920	32	68	376	230	0,40
27	8	26616,5	35	2,5	48,5	33980	35	68,5	428	235	0,36
37	8	26958	27	3,0	48	32130	37	67	439	220	0,43
47	8	26682	26	2	46	32570	34	59	488	180	0,31
57	8	28792	25	1,5	41	33330	38	58	536	190	0,29
59	8	28792	25	1,5	41	33330	38	58	536	190	0,29

	Lake Kozelská [mg/L]										
Depth [cm]	**Ag**	**Al**	**As**	**Cd**	**Cu**	**Fe**	**Ni**	**Pb**	**Ti**	**Zn**	**Hg**
7	8	22293	47	5,0	121	40780	45	106	515	620	2,82
17	13	30232	**97**	10	**203**	44190	56	220	667	**1550**	7,63
27	8	26446	**142**	12	**234**	55430	47	**302**	594	**1570**	7,69
37	8	28425	**126**	4,0	103	57090	41	218	544	730	2,59
47	**20**	26075	**91**	4,0	68	62990	36	122	639	300	0,64
57	8	14057	14	1,5	20	26350	25	26	318	80	0,11

<u>*Sediment pollution in the Elbe River floodplain*</u>

To get a wider context of the Elbe River sediment oxbow lake pollution, a comparison with previous studies was done. Mean concentrations of investigated elements in different oxbow lakes are shown in Table 29. Different color represents the class of I_{geo} in each sediment core using the background values determined by Prange et al. (1997). The level of pollution is then described in Table 30. The contents of determined elements in sediment of the investigated lakes were compared to the results from sediment research that was carried out in Lakes Němčice, Lžovice, Poděbrady, Václavka, Obříství (Chalupová, 2011), Labiště pod Opočínkem (Klouček, 2002) and Doleháj (Chalupová, 2003). Due to using aqua regia leaching, the concentrations were completely comparable.

Mean concentrations of metals depended on the length of sediment core. For this reason, exact lengths of collected cores from every lake are shown in Table 1. As the aim of the studies was to identify the contamination in the maximum sediment depth, the length of each sediment core differed due to individual sedimentation-erosion conditions in each part of the lakes.

Generally, the sediments of most of the selected oxbow lakes of the Elbe River showed high contamination load with silver and cadmium. In some lakes, higher pollution was also determined in cases of mercury and lead. The contents of nickel and iron showed the same class of contamination among all compared oxbow lakes. The sediments from Lake Kozelská contained higher concentrations of Cd in comparison to the other lakes.

Higher pollution of the sediments of the Elbe River oxbow lakes was recorded mainly near the significant industrial sources of contamination. The intensity of communication with the river also influenced the load of metals in sediment. The concentrations of determined elements in Lake Kozelská might be a result of its location near to Spolana Neratovice chemical plant. A significant sediment load was also found in Lake Labiště pod Opočínkem, which is located a few kilometers downstream Synthesia, a. S., In Pardubice – Semtín (Klouček, 2002).

Higher pollution with silver, cadmium and mercury was also recorded in the oxbow lake near Lžovice, located further downstream of Pardubice, which could be also caused by its intense hydrological connection with the Elbe River. Higher pollution by silver, cadmium, mercury and lead was also recorded in Lake Obříství, which is located several km downstream of Spolana Neratovice (Chalupová, 2011).

The lowest sediment contamination was identified in Lake Václavka, which was separated from the Elbe River already in the second half of 19th century (Krýžová, 2007), and it is not hydrological connected with the Elbe River by surface even during the 5-year flood (Chalupová, 2011).

The significant influence on the sediment load in the Elbe River oxbow lakes is therefore not only the distance from the source of contamination, but also the intensity of lakes' hydrological communication with the river.

		Mean concentration [mg/kg]										
Location	**Core lenght [cm]**	**Ag**	**Al**	**As**	**Cd**	**Cu**	**Fe**	**Ni**	**Pb**	**Ti**	**Zn**	**Hg**
Vrť 2017	59	8	26517	37	2	47	32816	35	67	452	221	0,39
Lake Kozelská 2017	57	12	24588	86	6	125	47805	42	166	546	808	3,58
Němčice 2007	67	2,3	-	20	0,8	61	944	31	76	-	478	0,44
Lžovice A 2007	151	11,2	-	20	4,6	209	890	38	89	-	563	3,99
Lžovice B 2007	103	8,5	-	20	2,2	97	900	33	84	-	557	2,66
Poděbrady 2007	204	2,5	-	37	1,8	85	912	34	96	-	483	1,8
Václavka 2007	67	0,4	-	20	0,2	58	912	30	50	-	310	1,17
Obříství A 2007	163	5,8	-	25	3,1	121	928	43	124	-	594	1,36
Obříství B 2007	187	1,6	-	22	1,6	79	936	29	79	-	629	3,41
Labiště 2002	50	11,4	-	-	2,93	85,8	17860	46	112	-	653	1,258
Doleháj A 2002	30	10,9	-	-	1	37,3	11523	35,8	99,6	-	206	0,405
Doleháj B 2002	15	2	-	-	0,5	35,7	20340	41,5	96,3	-	204	0,54
Doleháj C 2002	30	3,3	-	-	1,25	41,8	23060	41	108	-	239	0,155

Table 28: Mean concentrations of heavy metals and As in the Elbe River oxbow lakes with color differentiation according to Indices of Geoaccumulation (data source: Chalupová, 2011; Chalupová, 2003; Klouček 2002)

		Mean concentration [mg/kg]										
Location	Core lenght [cm]	Ag	Al	As	Cd	Cu	Fe	Ni	Pb	Ti	Zn	Hg
Vrť 2017	59	8	26517	37	2	47	32816	35	67	452	221	0,39
Lake Kozelská 2017	57	12	24588	86	6	125	47805	42	166	546	808	3,58
Němčice 2007	67	2,3	-	20	0,8	61	944	31	76	-	478	0,44
Lžovice A 2007	151	11,2	-	20	4,6	209	890	38	89	-	563	3,99
Lžovice B 2007	103	8,5	-	20	2,2	97	900	33	84	-	557	2,66
Poděbrady 2007	204	2,5	-	37	1,8	85	912	34	96	-	483	1,8
Václavka 2007	67	0,4	-	20	0,2	58	912	30	50	-	310	1,17
Obříství A 2007	163	5,8	-	25	3,1	121	928	43	124	-	594	1,36
Obříství B 2007	187	1,6	-	22	1,6	79	936	29	79	-	629	3,41
Labiště 2002	50	11,4	-	-	2,93	85,8	17860	46	112	-	653	1,258
Doleháj A 2002	30	10,9	-	-	1	37,3	11523	35,8	99,6	-	206	0,405
Doleháj B 2002	15	2	-	-	0,5	35,7	20340	41,5	96,3	-	204	0,54
Doleháj C 2002	30	3,3	-	-	1,25	41,8	23060	41	108	-	239	0,155

Table 29: Igeo classes of sediment pollution

I_{geo} value	I_{geo} class	Sediment pollution
≤ 0	0	unpolluted
≤ 1	1	unpolluted to moderately polluted
≤ 2	2	moderately polluted
≤ 3	3	moderately to strongly polluted
≤ 4	4	strongly polluted
≤ 5	5	strongly to extremely polluted
≥ 5	6	extremely polluted

9 Conclusion

The investigated lakes corresponded with their area to the character of oxbow lakes. The area of Lake Kozelská was above average and it was comparable to Lake Obříství. Lake Kozelská was also deeper probably because of its location in the lower course of the Elbe River, which is characterized by higher discharge.

As number of measures has been adopted and resulted in the improvement of water contamination from point sources of pollution since the 1990s, the contamination from non-point sources is still not very successfully solved. Thanks to this situation, the investigated lakes showed higher contents of $N\text{-}NO_3$. Water in Lake Kozelská and Vrť contained also the highest concentrations of N-NH_4 among the compared oxbow lakes in the middle course of the Elbe River. Lake Kozelská was also characterized by the highest average oxygen concentrations measured in July, when summer phytoplankton species developed. In Lake Vrť, higher conductivity values were determined in spring due to snow melting causing probably wash out of chlorides or other ions into the lake.

In general, water quality of the investigated lakes corresponded to the character of oxbow lakes influenced greatly by the river and human activities in the floodplain. In comparison to the Elbe River, they often showed lower content of N-NO3, on the other hand, the content of organic matter was higher as well as dissolved oxygen concentrations. Similar results were found also in the previous limnological studies carried out in the middle course of the Elbe River.

Concerning biota, *Arthropoda* was the most numerous phylum in Lake Kozelská Zooplankton composition corresponded to the abundancy of planktivorous fish (Hrbáček, 2000). In Vrť, phylum *Rotifera* showed the highest relative abundance in August, while *Arthropoda* dominated in October. Lake Vrť was also characterized by small area, which could influence zooplankton species composition too. One of the decisive factors was also the land use of nearby fields (Havlíková, 2011).

To sum up, all compared lakes represented mostly eutrophic aquatic ecosystems with zooplankton structure depended on the availability of nutrients (phytoplankton biomass), the character of each water body (shading), as well as the predation pressure of fish.

The investigated sediment samples were composed of homogeneous material without significant color changes. Their dark color indicated reduction conditions and the presence of organic matter. In most cases, the layers of sediment were classified as fine sand in Vrť and as medium silt in Lake Kozelská, which corresponded to the fluvial environment.

Sediment cores from both lakes showed a different distribution of heavy metals and As in sediment layers. In Vrť, the concentrations of almost all investigated elements were lower and showed only small differences of their content with the depth of the sediment core. In Lake Kozelská, a different situation was found. The concentrations of determined elements reached mostly higher values, and their content changed a lot with the depth of the sediment core. Although these changes could be influenced by dredging of sediments during communist regime or extreme floods in the locality, the high concentrations resulted probably from the situation of the lake about 2 km upstream from Spolana Neratovice, the chemical plant, which represented one of the biggest sources of pollution of the Elbe River in the past.

During extreme floods, suspended matter with adsorbed pollution could be transported even upstream, and then deposit in Lake Kozelská, which probably happened during the flood in 2002. Regarding different digestion methods, higher concentrations were found after total digestion than after aqua regia leaching.

Generally, the sediments of most of the selected oxbow lakes of the Elbe River showed high contamination load with silver and cadmium. In some lakes, higher pollution was also determined in cases of mercury and lead. As it was confirmed by previous studies, in Lake Kozelská, significantly higher contamination in comparison to the river was found, which resulted from old pollution of the locality with contaminated suspended matter that settled in the lake. However, even if Lake Vrť was connected with the river by surface, the sediment contamination was not so high, which could probably result from an absence of a near source of industrial pollution or thanks to the dilution of contamination by the Jizera River.

The contaminated sediments of the oxbow lakes coming from the old anthropogenic contamination of the Elbe River can be remobilized during floods, for that reason, the old loads can represent a secondary source of pollution. Under certain hydrological conditions or industrial accidents, pH or redox potential can change. The stable solid forms of toxic metals can be converted to soluble forms, which can contaminate the aquatic environment. These forms are easier to consume by living organisms and thus get into the food chain. The issue of contaminated riverbed sediments should be further studied, not only with regard to metals and arsenic pollution, but also for a number of other toxic organic and inorganic substances, which may represent a significant risk to the ecosystem.

10 References

Bábek, O., Faměra, M., Hilscherová, K., Kalvoda, J., Dobrovolný, P., Sedláček, J., Machát, J., & Holoubek, I. (2011). Geochemical traces of flood layers in the fluvial sedimentary archive; implications for contamination history analyses. Catena, 87, 281–290.

Balatka, B.; Kalvoda, J. (2006): Geomorfologické členění reliéfu Čech. Kartografie, Praha,79s.

Balatka. B. (1961): Podélný profil a poznámky ke genesi spodních a údolních teras středního Labe. SČSZ, 66, s. 6 - 22

Barron, J. J. and Ashton, C. (2007): The Effect of Temperature on Conductivity Measurement', Water, pp. 1–5.

Bencko, V.; Cikrt, M.; Lener, J. (1995): Toxické kovy v životním a pracovním prostředí. Grada, Praha, 282 s

Bordas, F.; Bourg, A. (2001): Effect of Solid/Liquid Ration on the Remobilization of Cu, Pb, Cd and Zn from Polluted River Sediments. Water, Air and Soil Pollution, 128, p. 391-400.

Borovec, Z. (1995): Zatížení sedimentů Labe a jeho přítoků toxickými prvky. Geografie - Sborník ČGS, 100, 4, s. 268-274.

Borovec, Z. (2000): Speciace prvků v kontaminovaných půdách, kalech, říčních a jezerních sedimentech. Vodní hospodářství, 50, 1, s. 1 - 5

Borovec, Z. (2001): Geochemical distribution of metals in aquatic sediments. Acta Universitattis Carolinae Environmentalica, 6, 1 - 2, p. 11 - 22.

Borovec, Z., Tolar, V., & Mraz, L. (1993). Distribution of some metals in sediments of the central part of the Labe (Elbe) River, Czech Republic. Ambio, 22(4), 200–205.

Broekaert, J. A. C.; Gücer, S.; Adams, F. (eds.) (1990): Metal Speciation in the Environment. Springer Verlag, Berlin, 656 pp.

Byappanahalli, M. N., D. A. Shively, M. B. Nevers, M. J. Sadowsky, and R. L. Whitman (2003):. Growth and survival of Escherichia coli and enterococci populations in the macro-alga Cladophora (Chlorophyta). FEMS Microbiol. Ecol. 46:203-211

Chalupová, D. (2003): Limnologické poměry, kvalita vody a sedimentů ve starém labském rameni Doleháj u Kolína. Diplomová práce. PřF UK, Praha, 102 s.

Chalupová, D. (2011): Chemismus vody a sedimentů fluviálních jezer Labe. Disertační práce. PřF UK, Praha.

Chlupáč, I. a kol. (2002): Geologická minulost České republiky. Academia, Praha, 436

Cullum, R. F. et al. (2006): Combined effects of best management practices on water quality in oxbow lakes from agricultural watersheds', Soil and Tillage Research, 90

ČHMÚ (2010B): ČHMÚ – Monitorovací programy. [online]. [cit. 2010-7-12]. < http://voda.chmi.cz >

Dědina, V. (1918): Příspěvek k poznání morfologického vývoje české tabule křídové. IV. Chlumecko. RČ A, 26, 25, 43 s.

Demek, J. (1987): Obecná geomorfologie. Academia, Praha, 480 s.

Dhagumdi,V. (2008): Dynamic performance of a submerged packed bed biological filter for wastewater treatment, SUNY-ESF, Syracuse, New York.

Engelmann, R. (1938): Der Elbedurchbruch. Geomorphologische Untersunchungen im oberen Elbegebiet. AGG, 13, 2, 139 S.

Fairbrodge, R.W. et al. (1968): The Encycopedia of Geomorphology. Reinhold Book Corp., New York, Amsterdam, London, 1188 pp

Forro, L., et al. (2008): A. Global diversity of cladocerans (Cladocera; Crustacea) in freshwater. Hydrobiologia. 2008-01-01, roč. 595, čís. 1, s. 177–184. Dostupné online [cit. 2017-08-16]. ISSN 0018-8158. DOI:10.1007/s10750-007-9013-5

Furnas, Miles J. (1990): In situ growth rates of marine phytoplankton: Approaches to measurement, community and species growth rates". Journal of Plankton Research. 12 (6): 1117–51. doi:10.1093/plankt/12.6.1117. INIST:5474600.

Goldman, Ch. R.; Horne, A. J. (1983): Limnology. McGraw-Hill, New York, 464 pp

Gordon, D. N. et al. (2004): Stream Hydrology. An Entroduction for Ecoogists. 2nd. Ed. John Wiley & Sons, Chichester,429pp

Graham et.al (2008): Blue-green algae in lakes and reservoirs—Toxin and taste-and-odor sampling guidelines' (ver. 1.0): U.S. Geological Survey Techniques of Water-Resources Investigations, book 9, chap. A7, section 7.5, September, available online only from http://pubs.water.usgs.gov/twri9A/.

Hach, C. (1997): Introduction to Biochemical Oxygen Demand. Technical Information Series – Booklet No. 7, USA, 22 p.

Havlíková, P (2011): Srovnávací studie fluviálních jezer středního Polabí horní Lužnice horní Svratky: [online]. Available from> https://is.cuni.cz/webapps/zzy/detail/84568.

Havlíková, P. (2007): Biologické hodnocení jakosti vody – srovnávací studie říčních toků a jezer fluviálního původu. Závěrečná zpráva GUAK č.321/2004/B-Geo. PřF UK, Praha, 75 s.

Havlíková, P. (2007): Biologické hodnocení jakosti vody – srovnávací studie říčních toků a jezer fluviálního původu. Závěrečná zpráva GUAK č.321/2004/B-Geo. PřF UK, Praha, 75 s.

Herben, T.; Münzbergová, Z. (2003): Zpracování geobotanických dat v příkladech. Část I: Data o druhovém složení. PřF UK, Praha, 118 s.

Hieftje, Gary; et al. (1982). "Design and Construction of a Low-Flow, Low-Power Torch for Inductively Coupled Plasma Spectrometry". Applied Spectroscopy. 36 (6): 627–631. Bibcode:1982ApSpe..36..627R. doi:10.1366/0003702824639105. Retrieved 5 April 2015.

Holz, D.; Pachur, H. J. (1992): Die subhydrische Sedimente der Groß-Glienicker Sees und ihre Kontamination mit Schwermetallen und einigen ausgewählten Umweltchemikalien. Berlin-Forschung, 11. Ausschreibung, FU Berlin, 60. S.

Horník, S. a kol. (1986): Fyzická geografie II. SPN, Praha, 319 s.

HRBÁČEK, J. (2000a): Zooplankton v pelagiálu a zarostlém litorálu tůně s rybím potěrem. In: PITHART, D. (ed.): Ekologie aluviálních tůní a říčních ramen. Sborník příspěvků z konference. Botanický ústav AVČR. Třeboň: 85-86.

Hrdinka, T. (2004): Antropogenní jezera České republiky. Diplomová práce. PřF UK, Praha, 115 s.

Jankowski, A. T.; Molenda, T.; Bebek, M.; Mitko, K. (2006): Zinc (Zn) and Copper (Cu) as Indicators of Bottom Deposits Anthropogenic Pollution. Limnological Review, Polish Limnological Society, Poznan6, p. 129-134.

Janský, B. (2002): Changing Water Quality in the Czech Part of the Elbe Catchment Aera in the 1990s (Twelve Years of Cooperation of Czechs and Germans on the Elbe River). Geografie - Sborník ČGS, 107, 2, p. 98-110

Janský, B. (2005): Nové trendy geografického výzkumu jezer v Česku. Geografie - Sborník ČGS, 110, 3, s. 129-140.

Janský, B.; Šobr, M. (2003): Jezera České Republiky. Monografie. PřF UK. Katedra fyzické geografie a geoekologie, Praha, 216 s.

Janský, B.; Šobr, M. (2004): Genetic Classification of Lakes in the Czech Republic. Geografie - Sborník ČGS, 109, 2, p. 117 - 128.

Jaworski, N. A., Howarth, R. W. and Hetling, L. J. (1997) 'Atmospheric deposition of nitrogen oxides onto the landscape contributes to coastal eutrophication in the Northeast United States', Environmental Science and Technology, 31(7), pp. 1995–2004.

Just, T. a kol. (2005): Vodohospodářské revitalizace a jejich uplatnění v ochraně před povodněmi. ZO ČSOP Hořovicko, Ekologické služby s.r.o., AOPK, MŽP, Praha, 359 s.

Kalff, J. (2002): Limnology. PrenticeHall, EnglewoodCliffs, 592 p.

Kettner (1954): Všeobecná geologie. Nakladatelství Československé akademie věd. 462, [1] s. Čs. akademie věd. Sekce geologicko-geografická

Klapper,H.(2002): Technologies for lake restoration, Papers from Bolse− na Conference . Residence time in lakes:Science, Mana− gement, Education J. Limnol., 62(Suppl. 1): 73−90, 2003.

Klimaszewski, M. (1978): Geomorfologia. Panstwowe wydawnictwo geologiczne, Warszawa, 1098 s.

Kliment Z.; Matoušková, M. (2008): Long-term Trends of Rainfall and Runoff Regime in Upper Otava River Basin. Soil and Water Research, 3, 3, s. 155 - 167.

Kliment, Z.; Kadlec J.; Langhammer, J. (2008): Evaluation of suspended load changes using AnnAGNPS and SWAT semi-empirical models. Catena, 73, 3, p. 286 - 299.

Kliment, Z.; Langhammer, J. (2007): Modelling of the erosion risk in the Blšanka river basin. In: Dostál, P., Langhammer, J. (eds.) Modelling natural environment and society. Nakladatelství P3K, Praha, p. 75 - 94.

Klouček V, Vaverová (2005): Lake restoration – Rekultivace eutrofizovaných nádrží metodou srážení fosforu hlinitými solemi [Lake restoration – Method of phosphorus chemical precipitation by using of aluminium salts]. Vodní hospodářství. 4: 97–98.

Klouček, O. (2003): Limnologické poměry, kvalita vody a sedimentů v Labišti pod Opočínkem. Diplomová práce. PřF UK, Praha, 86 s.

Komárek et al. (1992): Variability of some planktic gomphosphaerioid cyanoprokaryotes in northern lakes. – Nordic J. Bot., 12: 513-524.

Kořínek, V. a kol. (1987): Carp ponds of Central Europe'. In: Michael, Managed aquatic ecosystems, Elsevier, Amsterdam, p. 29 – 62.

Langhammer, J. (2004): Modelling the structural changes of water quality in the Elbe river basin. Ekologia, 23, 1, p. 157-169.

Langhammer, J. (2005B): Classification of the dynamics of water quality changes in the Elbe River basin. Journal of Hydrology and Hydromechanics, 53, 4, p. 205-218.

Langhammer, J. (2009): Water quality changes in the Elbe River Basin, Czech Republic, in the context of the post-socialist economic transition. GeoJournal, Springer. DOI: 10.1007/s10708-009-9292-7

Lellák, J., Kubíček, F. (1991): Hydrobiologie. Karolinum, Praha, 257 p.

Lichtfuß, R.; Brümmer, K. (1981): Natürlicher Gehalt und anthropogene Anreichung von Schwermetallen in den Sedimenten von Elbe, Eider, Trave and Schwentine. Catena, 8, S. 251-264.

Lichtfuß, R.; Brümmer, K. (1981): Natürlicher Gehalt und anthropogene Anreichung von Schwermetallen in den Sedimenten von Elbe, Eider, Trave and Schwentine. Catena, 8, S. 251-264.

Lochovský, P.; Zemanová, B. (1996B): Sledování těžkých kovů a arsenu v plaveninách Labe. Zpravodaj pro hydroanalytické laboratoře, 22, s. 7 - 13

Matoušková, M. (2005): Assessment of the human impact on the river network as a basis for the ecohydrological monitoring of streams. Geographical Review, CXXIX.,LIII., p. 35-46.

McCave, I. N.; R.J. Bryant; H. F. Cook; C. A. Coughanowr (July 1986). "Evaluation of a Laser-Diffraction-Size Analyzer For Use With Natural Sediments". Journal of Sedimentary Research. 56 (4): 561–564. Bibcode:1986JSedR..56..561M. doi:10.1306/212f89cc-2b24-11d7-8648000102c1865d. Retrieved 24 October 2013.

Merian, E.; Anke, M.; Ihnat, M.; Stoeppler, M. (eds.) (2004): Metals and Their Compounds in the Environment: Occurence, Analyses and Biological Relevance. Wiley - VCH, Weinheim, 1774 pp..

Miller,C. . (2000): Daphnia pulex (on-line). Animal Diversity Web. Accessed March 25, 2012 at https"//animaldiversity.ummz.umich.du/site/accounts/information/Daphnia_pukex.html

MKOL (2010): Mezinárodní program měření Labe [online]. [cit. 2010-06-06]. < http://www.ikse-mkol.org .>.

Mrňa, F. (1991): Užitá geochemie. Academia, Praha, 418 s

Müller, G. (1979): Schwermetalle in den sedimenten des Rheins – Veränderungen seit 1971 Umschau 24, str. 778 – 783

Němec, J.; Hladný, J. (2006): Voda v České republice. Consult, Praha, 253 s.

Ošmera, S. (1973): Annual cycle of zooplankton in backwaters of the flood area of the Dyje. In: HRBÁČEK, J., STRAŠKRABA, M. (eds.): Hydrobiological Studies 3. Academia, Praha: 219-253..

Oulehle, F. (2002): Limnologie a hydrochemismus v NPR Rejvíz. Diplomová práce. PřF UK, Praha, 62 s.

PETRÁNEK, Jan. On-line geologická encyklopedie [online]. 2007 [cit. 2016-12-27]. Dostupné online.

Pickering, R. (1981): Why study organic substances in water?. In: Greeson (ed.), Organic Substances in Water, Geological Survey Circular 848 – C, p. 3 – 5

Pilecká, M. (1997): Údolní meandry, jejich geneze a morfometrické charakteristiky. Bakalářská práce. PřF UK, Praha, 35 s

Pisarevsky, A. M., Polozova, I. P. and Hockridge, P. M. (2005):Chemical Oxygen Demand', Russian Journal of Applied Chemistry Translated from Zhurnal Prikladnoi Khimii Original Russian Text Copyright, 78(1), pp. 101–107.

Pitter, P (2015): Hydrochemie. Publishing VŠCHT, Praha, 792 pp.

range, A. et al. (1997A): Erfassung und Beurteilung der Belastung der Elbe mit Schadstoffen, Teilprojekt 2: Schwermetalle – Schwermetallspezies, Geogene Hintergrundwerte und zeitliche Belastungsentwicklung. GKSS, Geesthacht, 405 S.

Prange, A. et al. (1997B): Erfassung und Beurteilung der Belastung der Elbe mit Schadstoffen, Teilprojekt 2: Schwermetalle – Schwermetallspezies, Grafische Darstellung der Längsprofile - Filtrate, Schwebstoffe, Sedimente. GKSS, Geesthacht, 495 S. Prange, A. et al. (1997C): Erfassung und Beurteilung der Belastung der Elbe mit Schadstoffen, Teilprojekt 2: Schwermetalle – Schwermetallspezies, Zusammenfassende Aus- und Bewertung der Längsprofiluntersuchungen in der Elbe. GKSS, Geesthacht, 233 S.

Rudiš, M.; Valenta, P.; Nol, O. (2008): Effects of polluted sediments in flood plains on environment and ground water. VÚV T.G.M., Praha, 81 pp.

Sacca et al.(2012): Redescription of Rhiyodomus tagatzi (Cilliophora: Spirotrichea: Tintinnida), Based on morphology, ciliary pattern and small subunit ribosomal RNA. Gene sequence. Journal of Eukaryotic Microbiology, 59, 218-231.

Salomons, W. & Brils, J. (Ed.) 2004. Contaminated Sediments in European River Basins. – Report des European Sediment Research Network.

Schumm, S.A. (2005): River variability and complexity. Cambridge University Press, Cambridge, 220 pp

Šnajdr, M. (2002): Limnologické poměry, kvalita vody a sedimentů v mrtvém labském rameni u Obříství. Diplomová práce. PřF UK, Praha, 86 s.

Šobr, M. (2007): Jezera České republiky - Fyzickogeografické a fyzikálně limnologické poměry. Disertační práce. PřF UK, Praha, 235 s.

Sokol, R. (1912): Terasy středního Labe v Čechách. Rozpravy ČSAV, II.tř., 21, 32 s.

Švambera, V. (1939): Jezera na české straně Šumavy. Sborník ČSZ, Praha, 45, s. 15 – 23.

Száková, J. et al. (2004) 'Single-Purpose Atomic Absorption Spectrometer AMA-254 for Mercury Determination and its Performance in Analysis of Agricultural and Environmental Materials', Chemical Papers, 58(5), pp. 311–315.

Tolasz, R. a kol. (2007): Atlas podnebí Česka. ČHMÚ a Univerzita Palackého, Praha, Olomouc, 255 s.

Trejtnar (1978): Střední Labe. 1. vyd. SZN, 1978. 236, [iii] s.

Turek, M. (2004): Komplexní limnologická studie odstaveného starého ramene Libišská tůň v PR Černínovsko.Diplomová práce. PřF UK, Praha, 82 s.

Veselý, J.; Gürtlerová, P. (1996): Medieval Pollution of Fluvial Sediment in the Labe (Elbe) River, Bohemia. Věstník ČGÚ, 71, 1, s. 51–56.

Vitásek, F. (1958): Fysický zeměpis. Díl 2 Pevnina. Československá akademie věd, Praha, 603 s.

Vlnas, R. a kol. (2005): Hydrologická bilance množství a jakosti vod České republiky. ČHMÚ, Praha, 45 s.lang

Weiss (2016): MEANDERING RIVER DYNAMICs. University of Illinois at Urbana-Champaign. Urbana, Illinois. disertation

Wetzel, R. (1983): Limnology'.2nd ed. Fort Worth: Saunders. ISBN 0-03-057913-9

William J. Emery, Richard E. Thomson (2001). Data analysis methods in physical oceanography. Gulf Professional Publishing. p. 83. ISBN 978-0-444-50757-0. Retrieved 2011-02-06.

Žebera,K. (1956): Fluviální štěrkopísky na území speciální mapy Hradec Králové - Pardubice. Antrhropozoikum, 5, s. 381 - 384.

Žebera,K. (1956): Fluviální štěrkopísky na území speciální mapy Hradec Králové - Pardubice. Antrhropozoikum, 5, s. 381 - 384.

List of the Czech National Standards used:

ČSN 757221

ČSN EN 27888

ČSN ISO 10523 (75 7365)

ČSN EN ISO 9963-1 (757371)

ČSN EN ISO 11885 (757387)

ČSN EN ISO 8467 (757519)

ČSN EN 1899-2 (757517)

ČSN EN ISO15681-2

ČSN EN ISO 11732 (757454

List of shorcuts

ANC - acid neutralizing capacity

BOD - Biochemical oxygen demand

COD - Chemical oxygen demand

List of Tables

List of Figures

11 Appendix

Příloha 1: Bathymetric map of Vrť

Příloha 2: Bathymetric map of Lake Kozelská

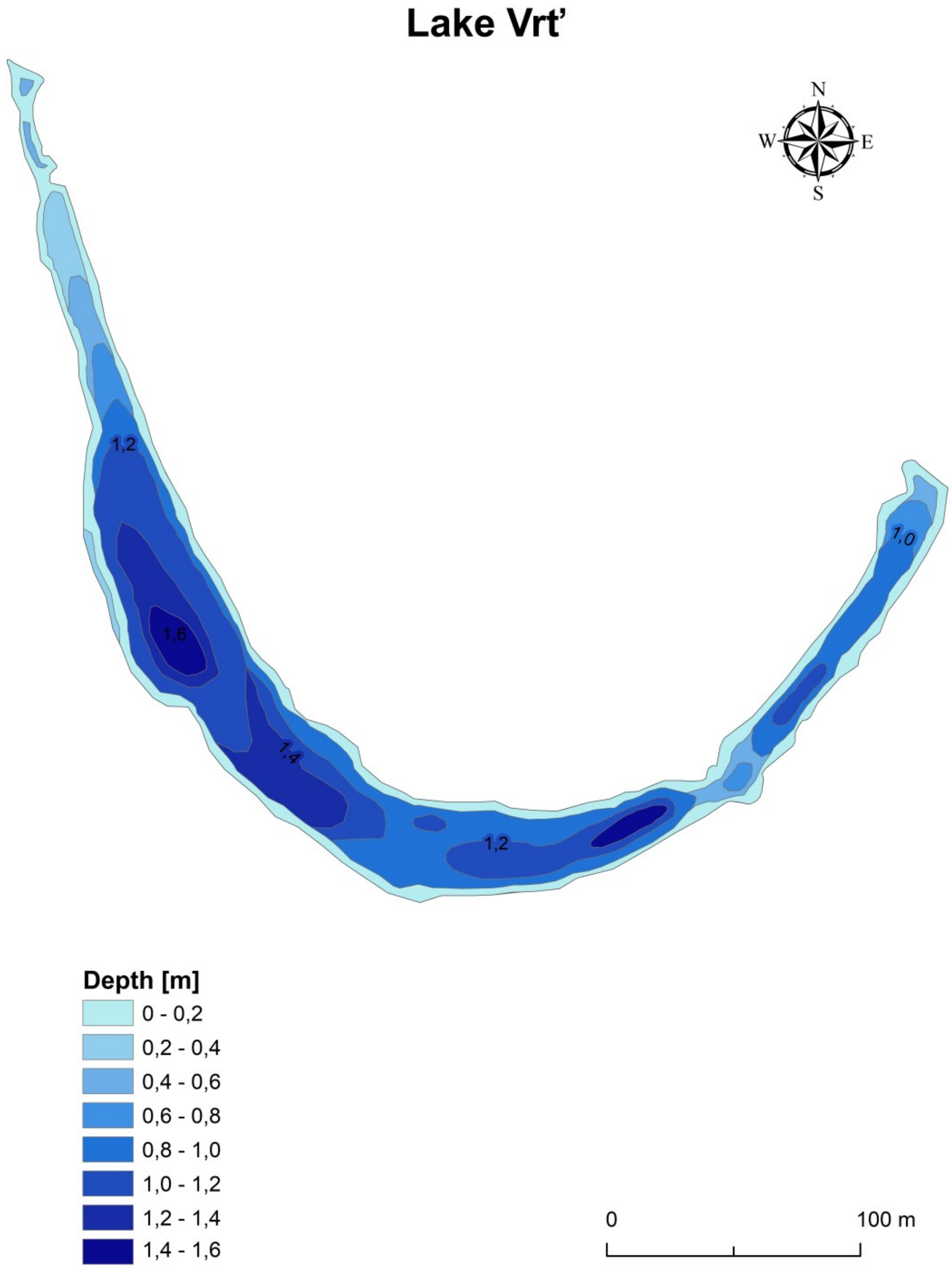

Figure 78: Bathymetric map of lake Vrť (modified from P.Havlíková, 2011)

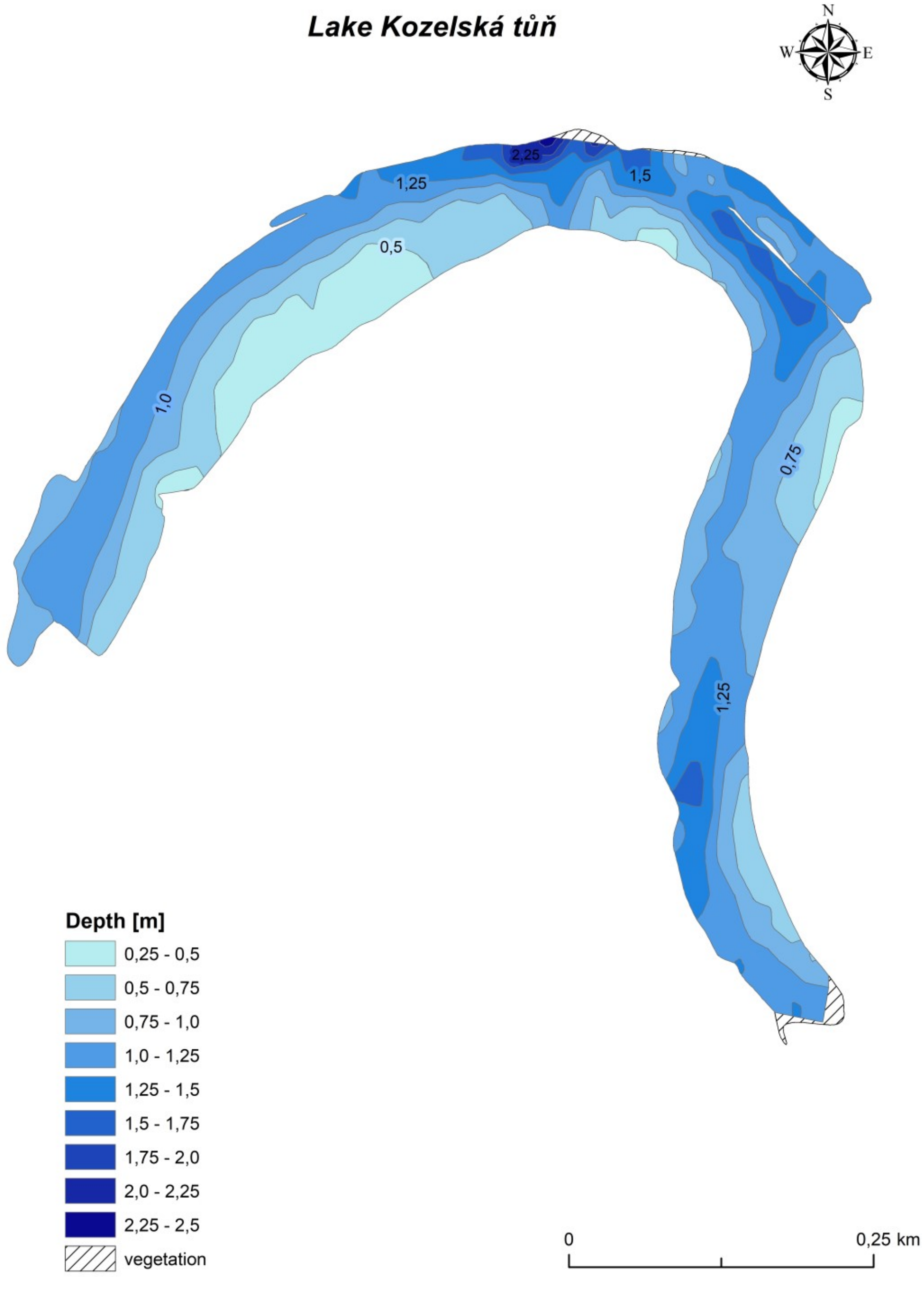

Figure 79: Bathymetric map of Lake Kozelská